A Manual on Mine Water Treatment and Management Practices in South Africa

APPENDIX
Volume 1
Literature Reviews

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WATER SAMPLING, SAMPLE PRESERVATION AND ANALYSIS INTERPRETATION

1 INTRODUCTION

The reliability of the results obtained from the analysis of water samples does not depend on the accuracy and precision of the analysis only; the sampling techniques, preservation and storage of samples are critical factors which are often neglected.

2 WATER SAMPLING

The success of an entire water-related project may depend on the collection of a representative sample. The primary objective of sampling is to obtain samples that are representative of the element that is to be analysed.

2.1 SAMPLING AND SAMPLE HANDLING

Water samples must be taken in clean containers (eg. bottles). The type of sample container and the cleanliness of the container requires special attention.

Glass containers are more easily cleaned but can break during transport and for that reason plastic or polyethylene containers are normally used.

If the sample bottle needs to be cleaned, the following method can be used :

- Soak the sample container in a phosphate-free **surfactant** for 24 hours.
- Wash it with water and thereafter with a dilute hydrochloric acid solution.
- Finally rinse three or four times with distilled water.
- The stopper should receive the same treatment as the container; contamination is often caused by inadequate cleaning of the stopper.
- Shake the container free of rinse water and allow to dry before the sample is taken.

The following aspects must be kept in mind when taking a water sample :

- The sample container must be rinsed, with a portion of the water to be sampled, before the container is filled.

- A sample taken from a running steam should be taken as near as possible to the centre of the stream.
- A sample should not be taken at a point where the bottom sediments have been disturbed.
- Before a sample is taken from a tap, the water should be allowed to run for several minutes, to avoid sampling stagnant water in the pipe and to ensure a representative sample and constant water quality.
- The mouth of the tap must also be washed free of contamination deposits formed by evaporation.
- Sample containers should always be filled completely, especially if chlorine, alkalinity, pH value, ferrous iron or nitrate/nitrite are to be determined.
- Filtration, if necessary, should be performed on site, and care should be taken to ensure that all equipment is as clean as possible.
- When sampling lakes, dams and reservoirs, the choice of sampling site must take into account the variation in parameters to be measured in relation to depth, season, wind, rain, etc. (1,2).

2.2 VARIABLES TO CONSIDER BEFORE PLANNING A SAMPLING PROGRAMME

2.2.1 Sample Mode

2.2.1.1 *Grab Sample*

This is used most often and the sample is taken by filling the container, this can be done manually or automatically. When taking the sample, unnecessary shaking and mixing with air must be minimised.

One way of doing this is to use a tube adapted to the sampling point and immerse it to the bottom of the bottle, to replace the contents of the bottle several times and to put the stopper on immediately. Avoid contact with air and shaking for analyses such as dissolved oxygen, carbon dioxide and pH, etc.

When daily variations in quantity and quality occur, a few grab samples are required as a single sample will be insufficient and not necessarily representative of the contaminant levels in the water.

Grab samples are also taken when unusual or undesirable situations are observed. For example, toxic residues such as cyanides are smelt or chromium or copper are observed by the colouring effect.

2.2.1.2 *Composite Sample*

A composite sample is collected over a period of time, i.e. 24 hour during which the same volume is sampled at time intervals and analysed as one. Time based sampling may be replaced by flow based sampling, where the sampling frequency is determined by flow and not set time intervals. This is normally done by automatic sampling apparatus but can also be done manually. The cost for chemical analyses can be reduced by compositing samples rather than analysing samples separately. Own discretion must be exercised in order to determine the validity of a composite sample against a series of separate samples. Separate samples may be analysed for a key contaminant and compared, to determine whether samples should be put together.

2.2.1.3 *Continuous Sampling*

It is often advantageous to know the variation in water quality during the course of a 24 hour period or even longer periods, in order to specify **diurnal** or seasonal variations. This is normally done by automatic samplers, where separate containers are filled during pre-programmed times of the day. Each sample is then analysed separately. The specific flowrate at the time of sampling is also very important information in order to determine the salt load passing through the system for that period.

Other logging equipment that may be used have probes in the water and save information in electronic format from where it can be down loaded to a PC. Data like electrical conductivity, pH and water level (flow) at a specific site can be accumulated in this way at a set frequency ranging from minutes to weeks over a long enough period to see influences of seasons, droughts and floods.

2.2.1.4 *Storm Flow Sampling*

Storms and thunder showers, that make up a very large proportion of the precipitation in the South African regions where mining occurs, often result in peak flows and flash floods. Apart from the amount of rain that falls over a limited period, the modification of the **geomorphology** of the local drainage **micro-topography** could also contribute to larger risks of raised run -off levels or floods.

There are a number of potential hazards associated with raised or storm flow (both in terms of safety and the environment) which motivate for storm flow monitoring to be a priority in the sampling procedure.

Many open cast coal mines in South Africa are situated in valleys and close to rivers which could lead to a flood hazard if the storm flow is not properly monitored and managed. Any shafts for platinum, gold or coal mining should also be protected from storm water, as water flooding into shafts is not only a cost to pump out again but could affect production and safety. In the literature reviews on underground water management and storm water management, these issues are discussed further.

Another storm water problem on mines is that the large slimes dam areas retain large volumes of rainfall, that the penstocks, return water canals and dams should be able to handle.

As storm water management is addressed in a separate literature review, only the factors influencing the sampling are presented here.

In terms of water sampling and analysis, intensive downpours probably have the most impact in terms of erosion of the side walls of slimes dams and increased seepage.

Many minerals are leached to the surface from the soil and slimes dam sludge by means of evaporation. Potential pollutants often concentrate on the surface layers as precipitates for example salts and metals like iron can form ferricrete-like layers on the surface of slimes dam sludge and iron oxides and hydroxides (yellow boy). These high concentrations of salts or minerals can be eroded and dissolved into water ways by intense rainfall events. These dissolved contaminants accompanied by larger suspended matter can influence sampling results during storm flow in various ways.

In general a tendency exists for initial run off to rinse the concentrated surface layers of soil or sludge, resulting in a high concentration of suspended and dissolved contaminants in the water. Depending on the source or properties of the contaminants, the high concentration of contaminants may persist in the storm water throughout the peak flow and thereafter, or else the peak flow may rinse the contaminants initially, followed by a dilution of the contaminants by the large volume of relatively clean water.

In order to obtain a representative sample of storm water, it is therefore often necessary to take samples directly before peak flow, during peak flow and when the peak flow subsides again.

2.2.1.5. *Microbiological Sampling*

Samples for bacteriological analyses are taken in sterile bottles, it is also important to make sure that sampling points such as taps are also sterile to prevent contamination. This is done with a flame before taking the sample. Sterile plastic bags are also commonly used for microbiological samples.

2.2.1.6. *Ground Water Sampling*

Just as flow is an important variable that needs to be noted for surface water sampling, the static pre-pumped water level of ground water should be recorded. The pump or extraction device used should be kept in mind when analysing for certain variables, especially the metals. The borehole needs to be **purged** and some variables (like surface water) need to be determined on site (temperature, EC, **Redox Potential Eh**, DO, pH and alkalinity). The pumping rate can then be checked and documented.

2.2.2 Size of the Sample

It is important to submit enough sample to the laboratory to conduct at least 2 to 3 analyses per determination, it is advisable to confirm the minimum amount required with the laboratory manager. If specific preservative measures are applied, it will be necessary to take separate samples for various analyses. All these factors will determine the size and number of samples to be taken.

2.2.3 Frequency of Sampling

The following factors may influence the frequency of sampling:

2.2.3.1 Day/Night Sampling

Daylight or the absence thereof affects **photosynthesis** of algae and aquatic plants which indirectly affects other variables. Sunlight also directly affects cyanide **photolysis**.

Photosynthesis in the presence of light produces oxygen and uses carbon dioxide. This affects dissolved oxygen concentrations and carbon dioxide related variables like alkalinity and pH. These in turn affect the **redox potential (Eh)** and solubility of contaminants like the heavy metals. **Autotrophic** aquatic plants that photosynthesise also use phosphorous and nutrients like phosphate and nitrate, affecting the sampling and analysis of samples.

2.2.3.2 Different Seasons

Rainfall and temperature are the two major climatological factors that affect seasonal changes. Additionally the day lengths could affect water quality due to photosynthesis and the reasons that have been discussed.

Rainfall causes major seasonal changes in water quality in most areas where mining occurs in South Africa. As discussed the rain can have a diluting effect or a concentrating effect, if erosion occurs.

Temperature affects the redox potential of water, the rate of chemical and metabolic reaction in the aquatic environment and furthermore directly influences certain variables like dissolved oxygen.

Most contaminants are more soluble at higher temperatures, but higher temperatures reduce the dissolved oxygen content of water.

Daylight periods that increase during summer, affect temperature and photosynthesis with the related implications for sampling.

2.2.3.3 *Wind*

Wind causes direct increases of dissolved oxygen in the upper layers of water bodies, and furthermore can cause nutrient circulation from the sediment.

Wind can cause characteristic lenthic (standing water) phenomena:

Oligomiktic, monomiktic, dimiktic, polimiktic and **meromiktic** lakes develop over time depending on the amount of mixing of the strata within the water bodies due to wind assisted by temperature differences (10).

2.2.3.4 *Rain*

Rainfall affects water quality indirectly through increased erosion or dilution, as discussed previously. The physical impact of the rain can however also increase the oxygen content of the water. Furthermore raindrops react with carbon dioxide in the air. This can lead to weak carbonic acid forming and could also affect carbonate related variables.

Various other contaminants like sulphur also react with raindrops to form acid rain that could also affect sampling and analysis.

2.2.3.5 *Depth of Sample Taken in Dam*

As discussed, temperature and oxygen content stratify standing water into layers that in turn may affect certain water quality variables sampled for.

Light penetration, oxygen dissolving and other reactions that occur closer to the surface, affect the upper layers of water bodies and sampling should preferably take place one third from the bottom of the water body if practical.

2.2.3.6 *Stopping and Starting of Various Industry Processes*

One factor that usually affects industrial processes and effluent discharge is the reduction of electricity cost due to lower demands during the late night and early morning hours.

Often sampling takes place during the day, while discharges might occur late at night, not giving a truly representative sample of the situation. Sampling over a 24 hour period and over a week often shows the events that influence the local water quality and the sampling program can be altered to incorporate the influence of discharges.

2.2.4 Location of Sampling Points

Location of sampling points within a water circuit or drainage network, is often subjective rather than completely random. It is often advisable to first sample a stream or water body at randomly set intervals along its length and then to sample additional sites where a site visit shows obvious potential impacts (positive and negative) that will affect the water quality at the selected sites. After an initial sampling survey, potential areas of importance may be identified and investigated in more detail.

After a specific set of data has been accumulated over time and compared, some sampling points may be re-located or discontinued, while new ones may be allocated.

2.3 STATISTICAL DESIGN OF A SAMPLING PROGRAM

An initial requirement for a data set obtained by a sampling program is for it to be compared to other data sets or specific requirements. Often in environmental monitoring it is necessary to determine the impact of a specific operation on a water body or stretch of river. In order to determine the impact, conditions upstream and downstream of the operation need to be monitored and compared. In the case of stagnant or slow moving water bodies, a control site, representative of back ground conditions in the area before the operation started needs to be identified and monitored (17).

Several statistical analysis techniques have been used to derive information from water quality data. By applying these techniques, recommendations may be made regarding screening of data and setting priorities for variable selection. The basic statistical analysis techniques that can be applied are regression, frequency distribution, time trend analysis, cluster analysis and regional patterns and distribution (11).

The statistics that are undertaken will be dependent on the amount of data available and the needs of the managers. Standard tests such as means, standard deviations, ranges and time series plots are recommended. Pictorial representation of the results enables easier understanding , e.g. box and whisker plots.

All raw data should be stored in a suitable spreadsheet or database (e.g. LOTUS, QUATTRO, EXCEL, PARADOX, HYDROCOM) on a personal computer. The personal computer should preferably be at least a 486-AT model with at least 100 MB of memory available and a maths co-processor to increase the efficiency of operation. At least one back-up copy should be kept of each data file, in a secure location and separated from the location where the data is routinely analysed.

A routine check should be made for the presence of possible computer viruses each time the data files are examined or manipulated.

Data analysis procedures should incorporate accepted routine statistical techniques for determining the mean, median, 95th percentiles and standard deviation values for each water quality variable at each sampling site.

Wherever possible, the data should be examined to assess whether or not there are any cyclical variations in effluent concentrations, as well as distinct changes with time.

In an analysis of possible concentration trends with time it is important to determine the seasonality of the data set. If a *distinct* seasonal cycle exists, the seasonal components of the data set should be removed before statistical analysis of any trends is conducted. This can be accomplished by using any one of a number of commercially available statistical software packages, *for example* SAS, SPSS, STATISTICA and STATGRAPHICS (11).

Once the seasonal components have been removed, the residual values should be examined to ensure that there is an even spread of values either side of the residual index. Subsequently, the seasonal components are removed from the original data set to produce a seasonally-adjusted data set. This seasonally adjusted data set is then plotted. With the seasonal components removed, it is now possible to evaluate any trend in concentration with time. This trend is then evaluated using a line/formula fitting technique to evaluate the rate of change with time. Where the rate of change with time is very variable, it may be more appropriate to split the deseasonalized data set into smaller (shorter) sets to evaluate each set of changes separately.

An auto-correlation analysis of the original data set can indicate whether or not there was auto-correlation between any data point and those collected at specified time intervals either before or afterwards. Taken on its own, this information will indicate what time interval between sample collection is required to obtain an independent sample where the concentration of a specific water quality variable recorded at one time is unlikely to be related to another data point.

If this exercise is repeated with the deseasonalized data set, a different picture could emerge. The auto-correlation analysis will reveal whether or not there are any cyclical changes in concentration, which increase the variability of the effluent quality. Such cyclical changes could; for example, be caused by a particular pattern of operations within the mine. Such an observation will allow management to develop and implement appropriate corrective measures to reduce this variability.

The most appropriate sampling frequency for compliance monitoring of an effluent stream will ultimately be determined by the particular water quality variable which requires the shortest time interval between samples. For example, in a case where one water quality variable should be sampled at weekly intervals and another at fortnightly intervals, it may be appropriate to sample both variables simultaneously at the shortest (weekly) time interval.

The second important component of a monitoring programme is the receiving water body. Again, similar seasonality analysis should be conducted for each water quality variable of concern as a basis for deciding on the most appropriate sampling interval and statistical treatment.

Subsequently, the seasonal components should once again be subtracted from the original data set and a seasonally adjusted data set plotted. With the seasonal components removed, it will now be possible to evaluate any trend in concentration with time. Once again, any trend analysis must consider whether or not the data set should be split into shorter sets to examine different portions of the pattern of change.

Against the background of the required frequency of monitoring for the effluent stream (for example, once per week), it would be inappropriate for compliance monitoring at the downstream river site to occur less frequently than that required for the effluent. Therefore, an overall sampling frequency may be selected that will reduce the expenditure of time and effort to a minimum and still provide reliable results(11).

3. PRESERVING SAMPLES FOR ANALYSIS

Once water is removed from the parent source, the composition of the water may change as a result of chemical or biological action. Chemical alteration may be caused by a change in **valence**, adsorption of metal **cations** by the container walls, precipitation and/or formation of stable complexes. To retard these changes preservatives can be added or as a general rule the sample is stored at 3 to 4°C. The following specific preservation treatment can be used :

- Add 1cm^3 of concentrated nitric acid per 100 cm^3 of sample to inhibit the precipitation of metals.

- For the determination of cyanide, the pH value of the sample should be adjusted to 10 - 12 with the addition of sodium hydroxide granules and the sample should be taken in a dark container that will not allow ultraviolet rays through.

- To determine phenolic compounds, adjust the pH value to less than 4 with phosphoric acid and also add 1g of copper sulphate.

From these preservation examples it is obvious that the preservative treatment will interfere with certain determinations. When planning a sampling program, separate samples may need to be taken for various analyses.

Table 1 indicates the primary preservation treatment of samples in terms of container to be used, preservation conditions and maximum holding period.

Notwithstanding the recommended maximum holding time, all analyses should be performed as soon as possible.

Table 1 : Sample Preservation

Determination	Sample Container	Recommended preservation conditions	Maximum holding period
pH	Pyrex	Analysis on-site preferably	24 hours
Hardness, M alk., Cl, Acidity, SO_4, colour, SS, dry solids	Plastic or Pyrex	Refrigeration at 4°C	48 hours
Tasting, threshold odour	Pyrex	Refrigeration at 4°C	Less than 24 hours
Dissolved oxygen	-	Determine on-site	No holding
PO_4	Plastic	40 mg/ℓ $HgCl_2$, refrigerated at 4°C and darkness	24 hours
CN^-	Pyrex	2 mℓ NaOH, pH 10, Refrigeration at 4°C and darkness	24 hours
Al, B	Plastic	5 mℓ HNO_3	4 weeks
Heavy metals, total	Plastic	5 mℓ/ℓ HNO_3	4 weeks
Metals, dissolved	Plastic	Filter : 3 mℓ/ℓ of 1:1/HNO_3	4 weeks
TOC (*Total organic carbon*)	Pyrex	2 mℓ H_2SO_4 , pH 2, 4°C	1 week
Hydrocarbons	Pyrex	2 mℓ H_2SO_4, pH 2, 4°C	24 hours
Phenols	Pyrex	1,5 mℓ H_2PO_4 and 10 mℓ 10% $CuSO_4$ solution	1 week
Bacteriology	*Sterile plastic* bottles/bags	Sterile flasks with thiosulphate Refrigeration at 4° C vital	24 hours
Nitrogen, Ammonia	Plastic or Pyrex	40 mg/ℓ $HgCl_2$ at 4°C	1 week
Nitrogen, Kjeldahl	Plastic or Pyrex	40 mg/ℓ $HgCl_2$ at 4°C	Unstable
Nitrogen, Nitrate	Plastic or Pyrex	40 mg/ℓ $HgCl_2$ at 4°C	1 week
Sulphide	Plastic or Pyrex	2 mℓ/ℓ Zn acetate	1 week
Conductivity, Fluoride	Plastic	None required	-
Oil and grease	Plastic	2 mℓ/ℓ H_2SO_4 at 4°C	24 hours
Mercury	Glass	-	4 weeks

4. ANALYSIS INTERPRETATION

4.1 THE ANALYTICAL REPORT

An example of a typical analytical report for a mine water sample is given in Table 2.

Most ions in the report are expressed in concentration by volume as milligrams per litre (mg/ℓ). For water chemistry the various types of *hardness* and alkalinity are often expressed in terms of calcium carbonate. This is to facilitate the calculation of an ionic balance which is a helpful tool.

4.2 CALCULATION OF THE IONIC BALANCE

From the analysis report the impurities can be categorised as cations (positive charge ion) and anions (negative charge ions). This serves a useful purpose when the concentrations for the anions as milligrams per litre are converted to milligrams per litre as $CaCO_3$. Adding up *the cations and anions separately* as $CaCO_3$, must balance if this is done for all the positive and negative ions present in the water. A difference between the two indicates a possible error in the analysis.

Table 2 : A Typical Water Analysis Report

Sample ID: .... Sample Date: .... Date Analysed: ....

Analysis	Unit	Result
pH		7,9
Conductivity	mS/m	-
Alkalinity at pH 4,5	mg/ℓ $CaCO_3$	110
Ca Hardness	mg/ℓ $CaCO_3$	150
Mg Hardness	mg/ℓ $CaCO_3$	180
Total Hardness	mg/ℓ $CaCO_3$	330
Total Dissolved Solids	mg/ℓ	765
Suspended Solids	mg/ℓ	16
Chloride as Cl^-	mg/ℓ	120
Sulphate as SO_4	mg/ℓ	310
Fluoride as F^-	mg/ℓ	0,5
Nitrate as N	mg/ℓ	<0,1
Orthophosphate as PO_4	mg/ℓ	-
Total Cyanide as CN	mg/ℓ	<0,1
Sodium as Na^+	mg/ℓ	105
Potassium as K^+	mg/ℓ	20
Langelier Index @ 25°C		1,27
Ryznar Index @ 25°C		5,36
Corrosivity Index		2,85
Total Aerobic Plate Count	cfu/mℓ	$3,5 \times 10^5$
Total Coliforms	cfu/100 mℓ	100
Escherichia Coli	cfu/100 mℓ	Nil

Table 3 shows the conversion factor from mg/ℓ to mg/ℓ as $CaCO_3$ for various ions.

Table 3 : Conversion Factors for Converting mg/ℓ to mg/ℓ as $CaCO_3$ for Different Cations and Anions.

Cations	Conversion Factor	Anions	Conversion Factor
Hydrogen	50,00	Hydroxide	2,94
Ammonium	2,78	Chloride	1,41
Sodium	2,18	Bicarbonate	0,82
Potassium	1,28	Nitrate	0,81
Magnesium	4,10	Bisulphate	0,52
Calcium	2,50	Carbonate	1,67
Ferrous	1,79	Sulphate	1,04
Ferric	2,69		
Cupric	1,57		
Zinc	1,53	**Other**	
Aluminium	5,55	Carbon dioxide	2,27
Chromic	2,89	Silica	1,67

Example: 60 mg/ℓ as Ca = 60 x 2,50 (Conversion factor)

= 150 mg/ℓ as $CaCO_3$

These factors can be calculated but are also available in water treatment handbooks (3,4).

The calculation for Ca for example is as follows (5):

Equivalent weight of Ca^{2+} $= \frac{\text{Molecular weight}}{\text{Valence}}$

$= \frac{40,1}{2}$

$= 20,05$

Equivalent weight of $CaCO_3$ $= \frac{\text{molecular weight}}{\text{Valence}}$

$= \frac{100,1}{2}$

$= 50,05$

Conversion factor for Ca as mg/ℓ Ca to mg/ℓ as $CaCO_3$

$= \frac{\text{equivalent weight of } CaCO_3}{\text{equivalent weight of Ca}}$

$= \frac{50,05}{20,05}$

$= 2,50$

If we would calculate the ionic balance for the water analysis in Table 2, we will end up with a set of results for the cations and anions as shown in Table 4.

Table 4 : Ionic Balance

	Result as mg/ℓ	**Conversion Factor**	**Result as mg/ℓ $CaCO_3$**
Cations			
Calcium	60	60 x 2,50	150
Magnesium	43	43 x 4,10	176
Sodium	105	105 x 2,18	229
Potassium	20	20 x 1,28	26
Total	228	-	581
Anions			
Hydrogen Carbonate	134	134 x 0,82	110
Sulphate	310	310 x 1,04	322
Chloride	120	120 x 1,41	169
Total	564	-	602
Total Salinity	792		Diff. 20

If the analyses are correct as well as the sampling procedures the total of cations expressed as mg/ℓ as $CaCO_3$ should be the same as that for the anions. Due to experimental error, the numbers will not be exactly the same (581 and 601), but should be within approximately 5% (6).

The total salinity (anions plus cations) in mg/ℓ should be of the same order as the TDS value (792 compared to 765).

4.3 SCALING INDICES

4.3.1 Langelier Saturation Index (LSI)

The Langelier Saturation Index (calcium carbonate saturation index) is a calculated number by which the corrosivity or scale-forming tendency of a particular water may be determined. A negative index indicates that the water is unsaturated and therefore corrosive, while a positive index shows that water is scale-forming (3).

4.3.2 Ryznar Stability Index (RSI)

Ryznar modified the Langelier Index to be more reliable in predicting the scale or corrosiveness of water(2). The following conditions are applied:

- If RSI indicates a value of 4 to 5, tendency is heavy scaling.
- If RSI indicates a value of 5 to 6, tendency is light scaling.
- If RSI indicates a value of 6 to 7, water is stable/equilibrium.
- If RSI indicates a value of 7 to 7,5, water is slightly corrosive.
- If RSI indicates a value of 7,5 to 8,5, water is very corrosive.

4.4 REPORTING OF NITROGEN COMPOUNDS AS N

It is sometimes useful to have the analytical results for all the nitrogen compounds reported as N mg/ℓ instead of say NO_3 mg/ℓ. This makes it easier to compare the results if reported in the same format.

The conversion factor is calculated with the molecular weights of nitrogen and the compounds. Table 5 shows the conversion factors.

Table 5 : Conversion Factors for Nitrogen Compounds

Nitrogen Compound (reported in mg/ℓ)	Conversion Factor (Divide to indicate value as N mg/ℓ)
NO_2	3,29
NO_3	4,43
NH_3	1,21
NH_4	1,29

Example: Nitrate reported as NO_3 = 100 mg/ℓ
Nitrate reported as N = 100 ÷ 4,43 = 22,6 mg/ℓ as N.

5. WATER QUALITY STANDARDS

With the analytical report for the water sample available, it is possible to determine whether the water is fit for use by different consumers and processes by comparing it to the relevant standard. These standards will be discussed in a separate literature review.

The following list gives an indication of the different standards and criteria available:

- SABS specification for drinking water
- Proposed drinking water criteria under consideration for application for South Africa
- COMRO guidelines for mine service water
- COMRO guidelines for hydro-power water
- COMRO guidelines for condenser cooling water
- General effluent standards
- Water quality criteria for steam generation
- Water quality criteria for irrigation
- Water quality criteria for rivers/dams (protection of aquatic life)
- Water quality criteria for livestock watering
- South African Water Quality Guidelines - DWA&F(12,13,14,15)

Water quality standards are addressed in a separate literature review and will therefore not be discussed in further detail.

6. COMPUTER PROGRAMS FOR CHEMICAL CONDITIONING OF WATER

6.1 INTRODUCTION

Various chemical modeling programs are available on the market which allow the user to determine chemical equilibrium conditions, characterisation of water, and the dosing requirements to obtain the desired water quality. Two such programs are called AQUACHEM and STASOFT. A third program called MINTEQA2 is a geochemical equilibrium specification model.

6.2 AQUACHEM

AQUACHEM is user-friendly interactive program (developed by COMRO) which allows chemical equilibrium calculations to be performed on mine waters. The parameters which can be handled are: calcium, magnesium, sodium, nitrate, ammonia, chloride, sulphate and alkalinity/acidity concentrations as well as pH. Allowance is also made for the effects of iron and manganese. Calcium carbonate and calcium sulphate scaling potentials, pyrite leaching, the effect of the addition of neutralising chemicals and the blending of any number of waters may be calculated. Some common chemical softening processes can also be simulated (7).

6.3 STASOFT

STASOFT is a program based on the chemical equilibrium model of the Ca-Mg-CO_3 system for low and medium salinity waters with a TDS less than about 1 000 mg/ℓ. The updated STASOFT Version 3.0 is probably acceptable for application to waters with a TDS level up to about 15 000 mg/ℓ.

The program is designed to provide solutions to a broad range of problems which the water chemist or engineer is likely to encounter in practice. These include the following:

- Calcium and magnesium softening and stabilisation.
- Blending of two or more waters with different chemical characteristics.
- pH adjustment of a water through chemical dosing.
- Effects of temperature and concentration changes on a water's characterisation.
- Evaluation of underground or treated waters when achieving two- or three-phase equilibrium.
- Treatment of calcium and H_2CO_3 alkalinity deficient waters.
- Stabilisation of waters to a specified calcium carbonate precipitation potential (8).

6.4 MINTEQA2/PRODEFA2 : A GEOCHEMICAL ASSESSMENT MODEL FOR ENVIRONMENTAL SYSTEMS

This is a geochemical equilibrium speciation model capable of completing equilibrium among the dissolved, adsorbed, solid and gas phases in an environmental setting. MINTEQA2 includes an extensive database of reliable thermodynamic data that is accessible to PRODEFA2, an interactive program designed to be executed prior to MINTEQA2 for the purpose of creating the required MINTEQA2 input file (9).

7. CONCLUSION

Water quality analysis can be expensive and time consuming, and if the proper procedures during sampling and analysis are not followed, the integrity of the data may be put in doubt. Sampling errors can therefore be costly and reduce the effectiveness of management decisions based on this data.

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POLICIES AND LEGISLATION APPLICABLE TO MINE WATER

1. INTRODUCTION

Although many companies are adopting environmental management policies of their own, legislation is still the prime driving force. Though world-wide water policies or legislation such as have been developed for environmental aspects like ozone depletion, greenhouse effects and bio-diversity, the legislation being developed are not practical, certain similarities are evident in the legislation being implemented in different countries.

This review will discuss the general policies, regulations and legislation in South Africa in comparison to Australia, Canada, USA and the UK. The different approaches to water resource management and regulation of mining activities will be discussed.

2. SOUTH AFRICAN LEGISLATION AND POLICIES RELEVANT TO WATER RESOURCES AND THE MINING INDUSTRY

The necessary powers to exercise control over the management of South Africa's water resources are vested in the Minister of Water Affairs and Forestry by the Water Act of 1956. Later amendments, notably the Water Amendment Act, 1984 (Act 96 of 1984), broadened water quality management. Industrial effluent, and sources other than effluent, e.g. water which arises as a by-product from industrial and mining activities and seepage or storm water runoff from a site, were made subject to pollution control regulations. The State was also given powers to counteract pollution before it takes place.

The Department has been reviewing its water quality management policy and strategies and has implemented a new approach to water pollution control which forms an integral and major part of water quality management. In implementing this approach far-reaching policy and strategy changes will be required and a number of technical investigations will be required to support the new approach.

The most significant environmental regulations in South Africa for the mining industry are: the Minerals Act (1991), the Water Act (1956), Atmospheric Pollution Prevention Act (1965), Environment Conservation Act (1989), Nuclear Energy Act (1993), Hazardous Substances Act (1973), Health Act (1977) and the Physical Planning Act (1991).

Environmental management in the mining industry is largely documented by the environmental management programme report ("EMPR"). The environmental management programme ("EMP") that forms part of the EMPR is legally binding in terms of the Minerals Act. The EMP should contain reference to the statutory requirements.

A new Chamber of Mines Guideline is being prepared which presents and discusses all the environmental regulations applicable to the mining industry.

2.1 WATER QUALITY MANAGEMENT POLICIES AND STRATEGIES

Far reaching economic, political, social and demographic changes throughout the Republic of South Africa have had major impacts on water quality. There has been a steady decline in water quality countrywide. As a result, the Department (referred to as DWAF) has embarked upon a programme to re-evaluate and develop its water quality management policies.

Salination of surface water resources and eutrophication present the most serious current water quality problems. Some of the groundwater resources have already been contaminated. Micropollutants, microbiological quality and sedimentation also warrant urgent attention.

DWAF is the custodian of a limited national resource, which has to be judiciously managed to ensure continued adequate water supplies of acceptable quality for all recognised users. From the quality point of view, recognised users as defined in the Water Act 1956 (Act 54 of 1956) are: domestic, industrial, agricultural, environmental and recreational.

Until recently, DWAF applied the Uniform Effluent Standards approach to water pollution control by enforcing compliance with the General and Special effluent standards. To counter continuing deterioration of water quality and to meet challenges of the future, Water Affairs changed to the Receiving Water Quality Objectives approach for non-hazardous substances and to the Pollution Prevention approach for hazardous substances. The Receiving Water Quality Objectives approach focuses on the fundamental water quality management goal, namely maintaining fitness for use.

To counter the limitations of this approach and to be consistent with environmental policy development world-wide, DAWF has decided to embody in its water quality management policy aspects of the anticipatory or precautionary principle to environmental protection. This principle encompasses all types of action to avert danger and minimise risk to the environment.

The consistent execution of this policy will require a hierarchy of water quality management goals. The hierarchy contains elements of DWAF's present policy but embodies the precautionary approach in the form of both source reduction and the application of minimum effluent standards. The hierarchy of the water quality management goals is as follows:

- Source reduction, recycling, detoxifying, and neutralisation of wastes will be encouraged. As mandatory standards for waste reduction or minimisation are not currently feasible, DWAF will promote voluntary action.

- If there is no alternative to the discharge of an effluent, the effluent must meet minimum standards, which may be either uniform or industry-related. The current General and Special Effluent Standards can probably be adapted to serve as minimum standards.

- If the *application of minimum effluent standards is not sufficient* to maintain the fitness for use of the receiving water body, then standards stricter than the minimum standards will be enforced. Such stricter standards will be site specific and will be based on the results of a waste load allocation investigation, in accordance with the Receiving Water Quality Objectives approach.

- Exemptions from compliance with the minimum effluent standards will be considered only as a last resort and only if the receiving water quality body has enough assimilative capacity (i.e. capacity to absorb the waste without it affecting its fitness for use). Relaxations would have to be justified on the basis of technological, physical, economical and socio-political considerations. To ensure consistency in the application of this hierarchy to individual cases, decision-making guidelines have to be developed. The need for policies and strategies to deal with groundwater pollution is clearly demonstrated. It is proposed by various authors that groundwater protection policies should range from maintaining groundwater quality to setting water quality objectives in accordance with the requirements of one of the water users. Cognisance must be taken of the following factors that fundamentally affect the development of water quality management strategies and policies:

  * Because South Africa is a water scarce region, reuse of effluents is a vitally important supplement to freshwater resources. Effluent, with the accompanying pollutants, must therefore be returned to the natural water bodies for reuse.

  * It is accepted that the water environment normally has a capacity, usually quantifiable, to assimilate non-hazardous pollutants without detriment to predetermined quality objectives. The assimilative capacity is part of the water resource and must be judiciously managed and equitably shared by all water users.

  * For those pollutants that are hazardous to the environment by virtue of their toxicity, persistence and extent of bio-accumulation, a precautionary approach should be adopted, aimed at minimising or preventing their entry into the water environment.

  * Water quality is highly variable. Many factors affect water quality and the resulting uncertainties introduce risks in decision making and therefore necessitate the interpretation of water quality in problematic terms.

- Increasing environmental awareness and consequent public involvement with issues concerning protection of the ecology demands that, whatever water quality management policy is adopted, it must be easy to understand, simple to operate and accessible so as to maintain *public confidence in the system*.

The principal categories of pollutants are listed below:

- Chemicals (acids, metals, salts, inorganic compounds and organic compounds)
- Suspended matter (particles and colloids suspended in water that may act as transport media for metals or pathogens)
- Pathogens including bacteria and viruses
- Thermal changes
- Radioactivity, if present in significant levels.

Specific pollution potential from the mining industry is highlighted by DWAF as being, *saline effluent discharge, acid mine drainage, metal pollution and* erosion. According to Section 23 of the Water Act it is an offence to pollute water by rendering it less fit for:

- *the purpose to which it could ordinarily* be put to use by other persons;
- the propagation of fish or other aquatic life;
- *recreation* or other legitimate purposes.

Fundamental to the Receiving Water Quality Objectives (RWQO) approach is the determination of the quantity and quality requirements of the different user sectors. Normally DWAF would depend on the user to state its requirements, but guidelines have been set by DWAF in 4 volumes.

The receiving Water Quality Objectives approach has so far been applied mainly where polluters have approached DWAF for relaxation of Specific or General Effluent Standards. Although the Receiving Water Quality Objectives approach may well result in site-specific standards, the experience so far has been that the investigations have in all cases resulted in relaxation of the requirements. There can be no doubt that the new approach has considerably facilitated decision making and has generally been favourably received both by industry and local authorities that have to comply and by those concerned with the protection of the environment.

Other strategies that affect water quality management can be summarised as; reconciling water demand and supply (water pollution control can not be divorced from water supply management), reconciling water quality protection with financial goals (trade-offs between water quality objectives and financial objectives need to be sought), command-and-control approach (water use and impact control by permit), public involvement in decision-making, the polluter pays principle (public responsibility may also apply), prosecution of offenders (DWAF prefers the co-operative rather than confrontational approach) and interpretation of water quality in probabilistic terms (95% of samples must be < standard).

2.2 SOUTH AFRICAN WATER QUALITY MANAGEMENT: EVOLUTION OF POLICY

Integration of water quality management with national environmental policies is important in South Africa. The Integrated Environmental Management (IEM) protocol developed by the South African Council for the Environment, incorporates the aquatic environment as an important resource to be managed and conserved.

The DWAF goal is to ensure the ongoing, equitable provision of adequate quantities and qualities of water to all competing users at acceptable degrees of risk and cost under changing conditions. Although economic merit is the dominant criterion for developing water projects, the water management strategy recognises that goals and services are important and it strives to identify these goals and allocate an appropriate significance to each(2).

A new procedure, termed scenario-based policy planning, makes it possible for decision makers to express value judgements by means of direct comparisons between specific alternative policy options, the consequences of which are specified as far as possible(3).

The approach adopted in water management strategies should follow guidelines laid down by the DWAF for integrated catchment and water quality situation assessments as the national policy is based on a catchment approach(4) and implementation of dynamic water quality management plans for different catchments.

The integrated catchment approach is based on;

- clearly delineated management units such as river catchments, subcatchments, groundwater compartments or total systems,
- setting of water quality management objectives
- development and implementation of wel designed strategies
- involvement of all role players in the planning process(5).

Industry/sector specific strategies form part of DWAF's overall water quality management strategy. The mining industry was the first sector for which such a strategy was completed and this is embodied in the EMPR. The implementation of this strategy will introduce a completely new and multifaceted approach to water quality management in this industry(6). IEM is a core aspect of the mining industry, and is leading South African industry towards a more environmentally aware business approach.

The receiving water quality objectives (RWQO's) approach and the precautionary principle, are fundamental to DWAF water quality management strategy, and thus also form the basis of the mining strategy.

The RWQO's approach is put into practice by dividing the country into logical functional entities usually established on a river catchment basis. The entities are termed management units. The setting of water quality objectives based on the user requirements (fitness for use) of users and the development of an associated management plan for a particular management unit is done on a site specific basis.

In order to ensure that the water quality objectives of a particular management unit are met, the collective impact on the water environment of all activities within the management unit must be compatible with the water quality management plan pertaining to the management unit. In essence this requires that the impact of each activity be quantified and evaluated in the context of the management plan. Mitigation will be required for those impacts not compatible with the plan.

The precautionary principle incorporates both the key elements of (i) *anticipation* of risk and (ii) *protection* against the degradation of the water environment. In terms of the protection element implementation of "best *practice*" based on the best available technology not entailing excessive cost (BATNEEC) will be enforced irrespective of whether impacts on the water environment are expected or not. The anticipating element necessitates up front insight into activities in order to minimise risks. IEM in mining typically fulfils this requirement.

DWAF's decision to focus on the mining industry stemmed mainly from the following:

- non-point sourced pollution is becoming increasingly important
- it is the single largest industry in South Africa
- it is perceived to be a developed, organised, and responsible industry which could assist in the development of the strategy; and
- the mining industry is already practising a measure of self-discipline regarding environmental matters(6).

The overall goal of the mining industry is to minimise the inevitable impacts of mining operations on the water environment by appropriate management action. In order to fulfil this goal the strategy embodies the following core aspects:

- IEM (Integrated Environmental Management)
- pecuniary provisions to cover environmental (water) liabilities;
- management of activities within the riverine environment

- more effective management of mine closures.

Other aspects addressed as part of the strategy include research guidance and co-ordination; legal matters; etc.

As DWAF is especially concerned with specific aspects associated with mining activities the above-mentioned take full cognisance of the following:

- surface disturbance insofar as water quality is affected;
- waste management;
- water management on the mining site; and
- other diffuse or point source discharges which could impact on surface and groundwater resources.

DWAF's interest in IEM in the mining industry lies in the fact that DWAF seeks to ensure that water quality issues are not omitted from the set of environmental consequences which are considered by a mine. Thus IEM in mining and the EMPR in particular, addresses water quality issues fully, but the submission of an EMPR does not absolve the proponent from complying with the requirements of the Water Act or those sections of the Environment Conservation Act for which DWAF has control. Such matters include, impacts on the water environment (Section 21), disturbance within the riverine environment (Section 20), water usage (Section 12), and waste disposal (Section 20 of the Environment Conservation Act).

It is thus foreseen that in most cases where water quality catchment management plans are in place no further information will be required for the process of the issuing of these permits. Where management plans are not in place more extensive impact assessments than specified in the Aide Memoire could be required from the proponent. The extent of this will have to be negotiated between the proponent and the department.

In the process of developing the EMPR, aspects which are of specific interest to DWAF include; demonstration by the mine that BATNEEC will be implemented irrespective of whether impacts on the water environment are expected or not; demonstration that the water resource related impacts associated with the mine's intended actions have been understood and that satisfactory mitigation measures are proposed in order to meet the required water quality objectives; demonstration that the above two aspects will be effected during the full life cycle of the mining venture.

DWAF is particularly interested in the demonstration of BATNEEC in the following areas:

- water management on the mining site;
- surface rehabilitation;
- waste management
- groundwater and surface water impact mitigation.

DWAF recognises the need to avoid duplication of environmental legislation and resulting duplications of approvals for mining projects. Thus the concept of a lead agent has been developed. Lead agent is defined as: "Any competent authority which assumes responsibility for the resource it is mandated to manage whilst serving the collective interested and affected parties to the satisfaction of the parties concerned." The Department of Mineral and Energy Affairs (from here on referred to as DM&EA) is best placed to serve the interests of all parties involved. Implementation of IEM and EMPR management procedures, interdepartmental agreements on procedure implementation, a forum for conflict resolution and communication, and the provision for pecuniary measures, auditing, monitoring and prosecutions, are some specific requirements in relation to the lead agent's role.

In 1990, before the Minerals Act of 1991, the uniform effluent standard was already in use to control pollution from point sources. The new approach which combines RWQO's and pollution prevention approaches, will apply to both point and non-point sources. The RWQO approach applies for control of non-hazardous pollutants and the pollution prevention approach applies to control input of dangerous hazardous substances(8).

DWAF is developing a holistic approach to water quality management and is in the process of reassessing goals and strategies to this end. DWAF's traditional role of controlling pollution and its place in the overall strategy is being evaluated. Some of the negative implications of the RWQO approach will be moderated by formally adopting aspects of a precautionary principle to water quality protection.

The overall goal must be to maintain the quality of the RSA's water resources such that it remains fit for recognised water users, within the framework of the major role that the reuse of effluent plays in the national water budget(7).

Incentive based measures for environmental protection include:

- Pollution charges, whereby a fee or tax is charged on the pollution load.
- marketable permit systems, whereby polluters have tradeable permits based on the assimilative capacity of the receiving environment.
- deposit-refund systems, where the consumer purchasing a potentially polluting product pays a surcharge or deposit.

3. RELEVANT SOUTH AFRICAN LEGISLATION

3.1. THE MINERALS ACT (ACT 50 OF 1991)

One object of this Act is to regulate the orderly utilisation and rehabilitation of the surface of land during and after prospecting and mining operations.

The right to prospect and authorisation to prospect is followed by the compliance with EMP's in EMPR's according to the Aide Memoire provided by the Department of Mineral and Energy Affairs. EMPR's are part of the IEM protocol of the Council for the Environment.

Discussion on underlying regulatory framework for EMPR's and IEM generally, states that no regulations on monitoring or auditing have yet been made (1).

3.2 THE WATER ACT (ACT No. 54, OF 1956)

This Act covers almost every aspect of the use of water, certain regulations were issued specifically to prevent pollution in the mining industry, as provided in section 22 of the Act prior to its amendment in 1984 (see R.287 dated 20 February 1976)(1).

Wilful or negligent pollution is an offence according to this Act, and the onus is on the accused to prove that an act that pollutes water was not wilful or due to negligence. The standard in this case is only that water is less fit for use after a pollution event.

A mine owner may remove from a mining area any water found underground if he is of the opinion that it is necessary for mining and safety of workers.

Water containing poisonous or injurious matter in suspension or solution must be effectively fenced off to prevent inadvertent access to it. Notice boards must be put up to warn persons not to use such water.

Rainwater management requirements include, erosion prevention of residue deposits, stormwater draining, provision for maximum precipitation in a 24 hour period and a freeboard of at least 0,5 metre throughout the system based on the expected maximum water level.

Permits are required for various actions like; using water for industrial purposes exceeding 150 cubic metres in any one day, erecting or enlarging water care works, using underground water for purposes other than the operation of the mine, or domestic purposes connected therewith, to use subterranean water in declared Subterranean Water Control Areas and lastly for the alteration of the course of a public stream.

3.3 THE ATMOSPHERIC POLLUTION PREVENTION ACT

The section on air pollution by smoke is relevant to coal mines where spontaneous combustion of discard dumps could lead to air pollution and indirect water pollution through acid rain, while pollution by dust fallout is a slimes dam problem.

3.4 THE ENVIRONMENT CONSERVATION ACT

Water use and disposal are listed among other activities that may have substantial detrimental effects on the environment. These activities are however well defined, in terms of regulations, under the Water Act.

Environmental impact reports are required to be compiled and submitted, by the act. Possible implications of proposed mining in sensitive environments, were well illustrated in the impact report for the mining of the Eastern Shores of St.Lucia. Regulations regarding waste, discarding, and dumping are covered by this Act.

3.5 THE NUCLEAR ENERGY ACT

This act came into force on 1 March 1994, and as water is one of the major pathways of radionuclides, this act is of importance, though most of the regulation focuses on solid material. The Hazardous Substance Act also addresses some radioactive materials.

3.6 THE HAZARDOUS SUBSTANCES ACT

Some chemicals used in the mining industry may be grouped in this Act, being toxic, corrosive etc. In terms of Group IV of this act, radioactive material which is outside a nuclear installation as defined in the Nuclear Energy Act, is also seen as a hazardous substance if it has an activity concentration of > 100 becquerels per gram or a total activity of > 4000 becquerels.

3.7 INTERIM CONSTITUTION

The Constitution of the RSA Act, No. 200 of 1993 ("The Interim Constitution") came into operation on 27 April 1994. Section 29 - Environment, of the constitution provides that any person shall have the right to an environment which is not detrimental to his or her health or well-being.

4. LEGAL & REGULATORY FRAMEWORK IN AUSTRALIA

4.1 NEW SOUTH WALES

4.1.1 Water pollution control

In 1992, the Environment Protection Agency (EPA) was formed under the provisions of the Protection of the Environment Administration Act 1991. The EPA is empowered by a number of pieces of legislation to act on behalf of the environment and the community. It's philosophy is that pollution prevention is better than control.

The approach of the EPA is to allow the polluters and users of water to consult between themselves to reach agreement on water quality issues. Should agreement not be reached then the EPA will intervene and the net result will be additional regulations which will complicate the issue further.

For water, most of the legislation rests with the Clean Waters Act of 1970. The act is designed to control pollution in rivers, streams, lakes, lagoons, natural or artificial water courses, dams, tidal waters and underground waters.

Water management systems which have been put in place in NSW, by the EPA, are similar in many respects to the systems currently being put in place in South Africa, in that the catchment as a whole is the basic unit of management. Where the major difference occurs, however, in Hunter Valley for example, is that unlike the South African catchment plans, the emphasis at the moment is only on salinity. No account is taken of other contaminants. Additionally, the Department of Water Resources has collected vast databases of water quality and flow information over time, and these are used for the determination of levels of salts permitted for discharge.

4.1.1.1 Case Study - Hunter Valley, NSW

The Hunter Valley has been divided into three discrete river reaches for water management. Additionally, the flow regimes of the rivers have been divided into low flow, high flow and flood flow. This division on flow regimes is a key concept in the philosophy of river management in NSW. Dilution is accepted as a fundamental water management tool, and in fact this embodies the concept of the right to pollute. The different river regimes, restrict the opportunity to utilise dilution to those times when there is sufficient flow. Low flow conditions will have little or no ability to dilute, therefore the focus of managing the system is aimed primarily on managing these low flows. The EPA will be phasing out and disallowing discharges during low flow periods.

The whole concept of pollution control has been investigated by the EPA. Since its emphasis of management is on salinity, much work has been done on identifying the sources of salinity. To date, the following salinity sources have been identified:

- natural salinity from the marine sediments which over time has leached into the surface and ground water systems.
- irrigation return flows
- coal mines

In terms of coal mining, periodic discharges of high salinity water, to a river which has a low salinity background, causes a spike. Attention is therefore focused on this spike of water as opposed to the increase in background concentrations. Over time, and with passive discharges (such as seepage) the background concentrations will rise, and this would pose a longer term impact. In order to address the issue of the spikes, a fundamental component of pollution control from mines, is that storage for water must be provided such that all effluents can be stored on site, until the flow in the river can accommodate a release without detrimental effect. The result is extremely large effluent reservoirs. It is not expected, however, that the water be stored indefinitely. Where there is inadequate land for storage, the marginal costs rise considerably. Under emergency conditions, approval to discharge can be obtained from the Minister for the Environment, but this is seen as unsatisfactory as the timing of the release may not coincide with optimal flow and salinity conditions.

4.1.1.2 *Disposal options for coal mining*

In order to dispose of the water stored by the mines, a number of options are available.

Transport by pipeline : A pipeline could be constructed to take the water from the upper reaches of the river to the estuary. As expected the cost of such a pipeline would be prohibitive;

Deep Well Injection : Water could be injected under pressure to deep permeable aquifers. In order to cope with the volumes of water from the mines, some 7000 boreholes would be required. This is not practically possible, nor economically viable.

Evaporation : While this can be used in association with other management practices, the volumes of water for evaporation are too great to rely solely on this method.

Desalination : Desalination, by reverse osmosis, is expensive, but does produce a good quality water at the end. One of the problems is the concentrated brine as waste.

Staged Discharges : In 1992 it was estimated that by 1995, some 4 600 Mℓ of water would need to be discharged in an average rainfall year, and 8 600 Mℓ for a wet year. This would add approximately 47 % and 12 % to the salt load respectively. There is however variation within the year in terms of flow. By taking advantage of times when river conditions are favourable, mine water discharges can be made with considerably reduced impacts on the river systems. This appears to be the most cost-effective manner for disposing of mine waters.

Licenses to discharge specify both an absolute and an incremental limit on the impact of mine water discharges on river salinity levels. The absolute limit is 420 mg/ℓ and the incremental limit is 24 mg/ℓ. Because of the incremental limit on existing licenses however, the discharge rate drops for staged discharges, to 1 % of river flows, or 1500 Mℓ/annum. This is obviously well below the required quantity.

Exceptional discharges : Another solution is to allow coal mines to discharge excess water in unlimited quantities at certain times of the year. This would tend to coincide with high flow (and therefore high rainfall events) and with low demand for water for irrigation (because of the high rainfall). To this end, a number of mines have developed sophisticated monitoring networks, up to 100 km downstream. These networks, which use telemetry to send the data from the individual stations to the mine, allow computer programmes to determine when the optimum conditions for release are approaching. Computerised systems then take care of the discharge of water from the storage reservoirs when the floods are moving past. These reservoirs are designed to enable rapid discharge of water. This enables the mine to undertake better planning because it can design the storage reservoirs according to the predicted flood return period.

Reservoir releases : This option uses the same principle as exceptional discharges, but uses artificial releases from impoundments to simulate the flood conditions. In areas and times when water is freely available, this could be an option. Where water is scarce however, the reduction in overall yield from the reservoir may affect other users of the water.

Tradeable permit schemes : One economic tool is the use of tradeable permits.

"The aim of introducing trading schemes for pollution abatement is to ensure that the operation of industries as a group (in this case coal mines) is both cost effective and environmentally effective. There must be incentives for the coal mines to make decisions that minimise overall production costs, including water management and other pollution abatement costs"(2). Additionally, decisions based on impacts which entail capital works, lock up funds in specific water management systems. The aspect of tradeable permits is currently being implemented and evaluated in Hunter Valley. The principle is that an acceptable salt load for the river is divided into a number of units and held in a salinity bank.

The nature of water flows and salinity content of the rivers needs to be brought into the equation in terms of tradeable permits. During certain flow conditions some (but not all) discharges can be accommodated in the river. The question then is who can discharge. If there were a set number of permits to discharge, one option would be to review the results from the monitoring programme daily, and apportion the discharge load accordingly. When the conditions in the river are unsuitable, no discharge would be allowed. When the conditions were right, those mines which need to discharge excess amounts could trade with other mines or polluters which did not need to discharge the full permit allowance, in order to be able to discharge excess water. Since all the polluters have the ability to purchase these units in order to discharge their effluent, those operations with good water management practices in place would buy proportionally less than those operations with a poor water management system.

4.1.1.3 Applicability to South Africa

The applicability of such systems to South African conditions varies.

- Pipelines and deep well injections are not feasible because of the practical and economic issues.
- Evaporation is practised extensively
- Desalination is expensive

- Staged, exceptional and artificial releases have some potential. Although the whole aspect of dilution is contentious, South Africa does periodically have extreme flood conditions arising. If the monitoring systems were in place, then releases could be made at these times. One important factor which must be taken into account, is the fact that all of our rivers are heavily impounded. If the flood is not sufficient to flush downstream dams, then the net result would be to dump the salt load into these dams, thereby affecting the use of the water for other purposes.

- The issue of tradeable permits would have to be investigated in some detail. In South Africa, the current strategy of catchment management looks at a variety of contaminants and the synergistic effects of one contaminant on another makes the issue of tradeable permits extremely complex. Additionally, the receiving water quality objective incorporates an element of assimilative capacity of the river, which applies to certain sections of the river. Any tradeable system that is applied to the whole river would have to take the assimilative capacity of each river reach in account. This strategy could have a potential to succeed in certain river systems, where single contaminant problems occur.

4.1.2 Mining Environmental Regulations

The process which is followed to get a mining authorisation is presented in Figure 1.

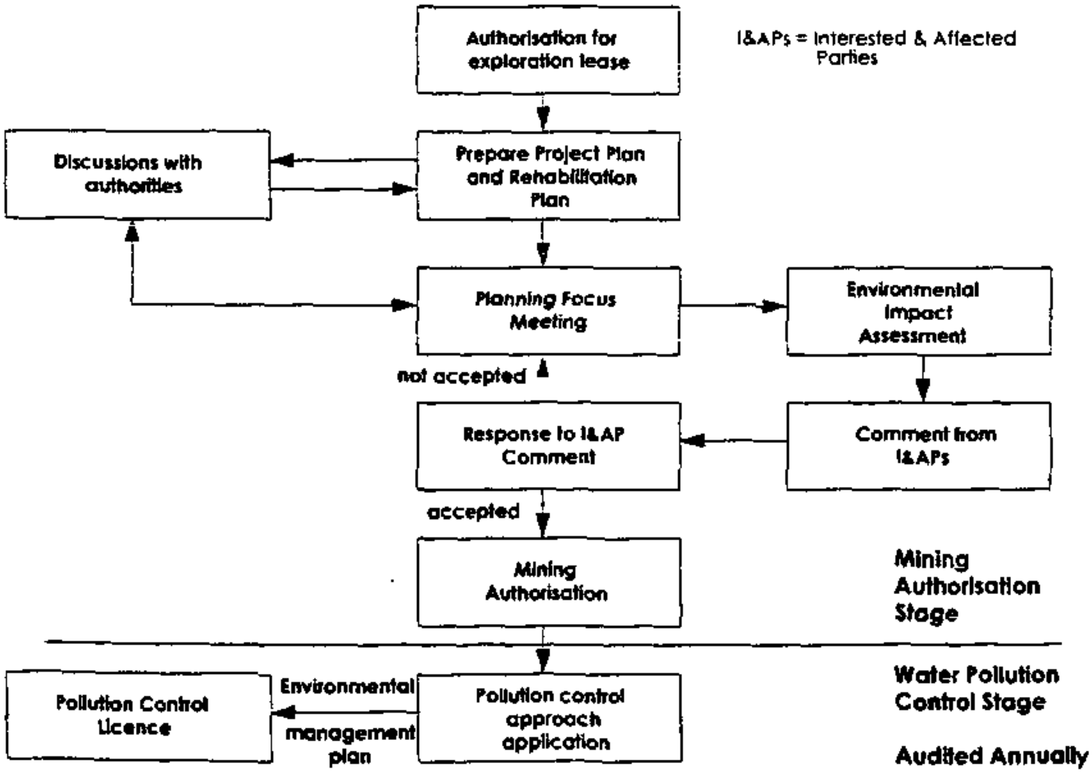

Figure 1 : New South Wales Mining Application Process

The process followed involves extensive discussions with the authorities and a significant involvement of I&APs. A detailed impact assessment is required which is scrutinised by authorities and I&APs. Once all parties are satisfied that the impacts have been addressed satisfactorily, the mine enters into the water pollution control phase. Under this process, application is made to obtain a pollution control licence. This licence is reviewed annually.

The interesting aspect for NSW, is that there is no attempt to impinge on the right to pollute. The only consideration is that it is done at a time when the base flow in the river can accommodate such a discharge without negatively affecting downstream users.

4.2 QUEENSLAND

4.2.1 Water pollution control

There are a number of regulatory authorities in Queensland, which have a stake in the environmental water management operations relating to the mining industry. Firstly, the Department of Mineral and Energy controls all aspects of mining rehabilitation. Discharge licenses for disposing of water, rest with the Department of Environment and Heritage. Currently the basis of regulation lies with the Clean Waters Act (1972). All pollution control dams on the mines, are under the control of the Department of Primary Industries' Water Resources Commission.

The new Environmental Protection Act, which went to Parliament in late 1994, will devolve all control for mining operations to the Department of Mineral and Energy.

In terms of discharge of polluted water, the same rationale for exceptional discharges applies. Sophisticated monitoring systems feed data to the mine and a central computer, which determines the conditions for discharge. These monitoring stations are maintained by the Water Resources Commission and measure flow, pH and conductivity. Discharges are only permitted during certain rainfall events. For example, at Collinsville, three rivers (at monitoring points up to 100 km away) must each have flows exceeding 0,1 m^3/second, 11 m^3/s and 110 m^3/s respectively, before Collinsville Coal can discharge.

4.2.2 Mining Environmental Regulations

The Department of Mineral Resource Industries has adopted a policy that mineral development licenses and mining leases be managed in a responsible manner in order to achieve in the long term, the self-regulation of the industry. Mining and rehabilitation should aim to create a land form which is the same after mining as prior to mining, the land should be self-sustaining (or at least where maintenance requirements are consistent with the post mining land use) and that no surface or ground waters leave the leased area to the detriment of downstream users.

The process followed is briefly described below and presented schematically in Figure 2. Mines must prepare an Environmental Management Overview Strategy (EMOS). This document contains broad principles and approaches with regard to environmental management at the mine. The EMOS process was initially developed in consultation between the mining industry and the authorities, in the same way the EMPR was developed in South Africa. The main differences to the EMPR however, are that there is no detailed environmental impact assessment, only broad strategies are supplied and the document is approved by only one authority, the Department of Mineral & Energy.

This EMOS gives rise to a Plan of Operations which gives a scheduled detailed plan of action for all environmental aspects. This plan also presents the financial security calculated for the rehabilitation of the mine over various phases of operations. This is reviewed annually against specific performance criteria and if the mine is found to be lacking, then it is placed in a different category.

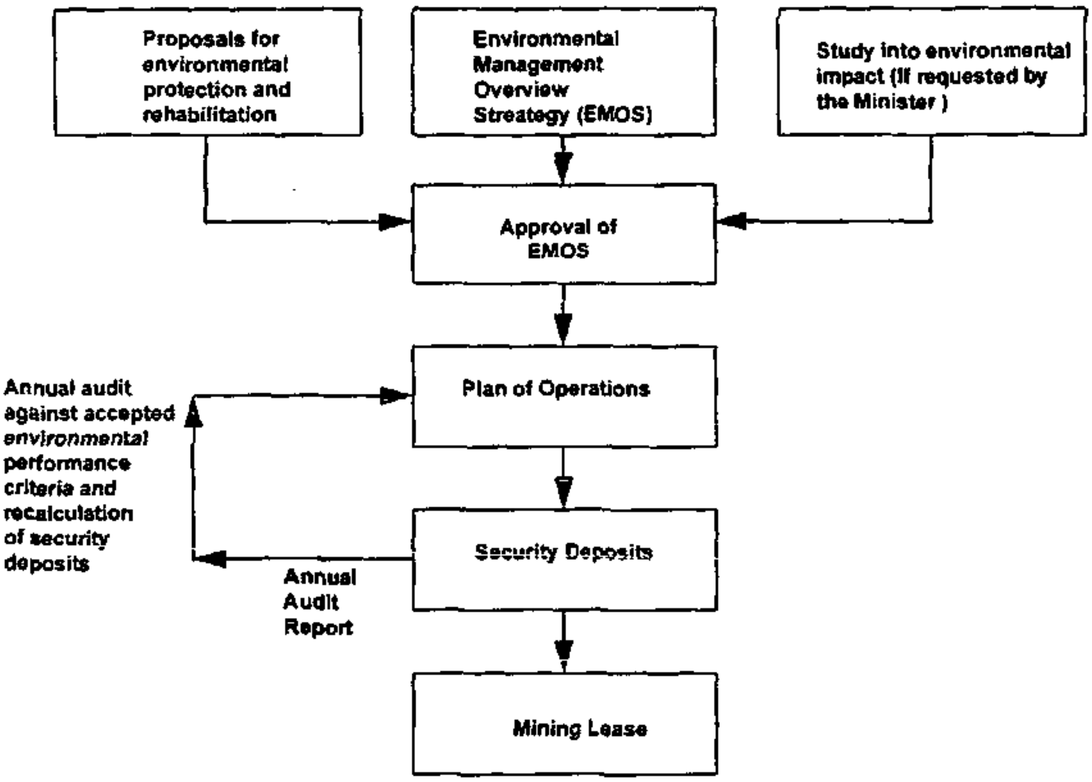

Figure 2 : Queensland Mining Approval Process

These categories relate to environmental performance and base the security deposit on percentage of costs to rehabilitate. Class 1 operations pay over 25 percent, and class 6 operations 100 per cent. The calculation of the amount of security to be deposited is based on the following factors:

- compliance with lease conditions and the provisions of the Act;
- the nature and extent of the actual damage the Crown may have to rectify; and

- the exposure to risk that the Crown may have to undertake this rectification.

The mine itself initially calculates the deposit and presents this schedule with the Plan of Operations. The net result of the security is to reduce the risk to the Crown for rehabilitating the mine should the lease holder default.

Typical aspects which are included in the calculation of the security deposit are the following:

- the maximum estimated area of land not rehabilitated under the current Plan of Operations;
- the actual cost of the rehabilitation; and
- the environmental performance category of the lease holder at the time of lodgement.

The environmental performance of the mine is audited annually. The proposed security deposit is calculated and an audited environmental document is prepared. This document includes:

- the EMOS
- the Plan of Operations
- the Environmental Performance Criteria

The audit report prepared must include an itemised list of EMOS programmes and in parallel with these, the associated action plan and audit comment for each. It shall also show the performance category assessment, the cost of rehabilitation and the security to be lodged.

Based on the environmental performance, a mine may move up or down the category list, or remain the same. The benefit is obviously to move up the scale to a Category 1. As an indication of the type of environmental awareness required, a Category 4 mine (i.e. security of 70 % required) must employ a dedicated environmental manager.

Security can be paid in one of three ways :

1. the money is paid directly to the state
2. the mine provides a bank guarantee
3. the mine obtains a credit rating from the state.

South Africa does not have a system of security deposits required for rehabilitation, although there are some indications that it is being investigated. This method of ensuring rehabilitation by way of proportional payments up front holds promise in South Africa, in that those companies who already apply sound management practices would get benefit, and those which do not, would be penalised.

5. LEGAL & REGULATORY FRAMEWORK IN THE UNITED STATES OF AMERICA

5.1 WATER POLLUTION REGULATIONS

In the USA, both Federal and State regulations control environmental issues. State limits can only be equal to or more restrictive than Federal limits. Current legislation for water pollution rests with the Federal Water Pollution Control Act, or as it has become known, the Clean Water Act. Since its enactment in 1972, it has been reviewed twice: once in 1986, and again in 1992. Apart from the two total revisions to the Act in 1986 and 1992, some amendments were also made in 1980, 1981 and 1987 where the Senate Environment and Public Works Committee embarked on re-authorising the Clean Water Act to include key issues. Provisions may include the following; storm water programs, agricultural non-point source pollution, enhanced pollution prevention funding, permit fees and, enforcement. Pollution prevention and waste minimisation are new priorities not only for federal programs but for corporate programs as well(10).

In the USA, the Water Quality 2000 coalition was founded to provide a new initiative in policy making. Initiated and administered by the Water Pollution Control Federation, the coalition brings together 86 organisations across a broad spectrum of opinion, from government and industry to academics and professionals. Reflecting the holistic approach to pollution problems advocated by Water Quality 2000, it is anticipated that changes may even be required in the administrative structure responsible for carrying out policy(11).

When the Clean Water Act was passed, provision was made for the States to develop area-wide plans for water management. But because it was not obligatory, the so-called watershed approach has rarely been adopted. Ideally, one plan would be devised and all the resources audited for an entire drainage area, regardless of boundaries. The enforcement of an area-wide plan is inhibited by the fact that the agreement is not binding, and its success depends on the goodwill of the participating states. The enforcement of federal regulations is also impeded by the lack of regional water authorities.

The Clean Water Act of the USA, in conjunction with other regulatory requirements that need to be met, affect mining operations directly. Apart from the Clean Water Act, the legal framework includes river use in the USA.

Use and development of rivers takes place within a complex legal framework that guides the actions of interested parties holding conflicting values. One of the major components of this legal framework is state water allocation law. This law takes several forms among the states but its basic function is to establish rules for water use by assigning property rights in water. Another component of the legal framework for river use addresses the public access issue. The primary focus here is establishment of the boundary between the exclusive rights of landowners and the rights of the general public to use the surface and underlying beds of waterways. A third component consists of governmental controls over water use and development. These controls restrict the exercise of private property rights in the resource. From an initial focus of protecting the public right of navigation, this law has expanded to include protection of water quality and a variety of other environmental values. The current legal framework provides a relatively sound basis for balancing competing values and interests, but problems exist. For example, the division of water management responsibilities between the federal and state levels of government sometimes results in conflict, and existing procedures for managing conflict related to proposals for water transfer do not appear to be adequate. Therefore, the current framework will not remain static; it must continue to evolve in order to operate more effectively and to incorporate future increases in knowledge and the associated changes in values and management needs.

Human use of rivers takes place within a complex set of physical and biological processes, in addition, use must take place within an intricate framework of institutional processes created by humans. Understanding these institutions sometimes seems more difficult than understanding complex natural processes, but water resources users and managers must operate within the institutional framework and recognise the constraints that are imposed.

The approach adopted is to divide the subject into three components: water allocation law (which defines property rights in water), public recreational rights (which allow non-consumptive uses such as boating and fishing), and governmental regulation of development (which omits the exercise of property rights in water by restricting activities such as water-related construction).

Permits to discharge water are comprehensive and encompass a variety of chemical and physical constituents. Water actively or passively discharged from mines is regulated by the National Pollutant Discharge Elimination System (NPDES). In terms of these regulations 4 elements are regulated, pH, total iron, total manganese and suspended solids. The specific values are recorded in Table 1.

Table 1 : NPDES Regulations

Element	**Daily maximum**	**Monthly average**
pH	6 - 9	no standard
Iron	6 mg/ℓ	3 mg/ℓ
Manganese	4 mg/ℓ	2 mg/ℓ
Suspended Solids	70 mg/ℓ	35 mg/ℓ

In addition to the federal limits as set out in the NPDES, most States have applied additional effluent regulations which are more stringent and which may cover a wide range of variables such as arsenic, specific metals, temperature, etc.

5.2 MINING ENVIRONMENTAL REGULATIONS.

There are two *different regulation* systems applicable to mining. Hard rock mining (including gold mining) is registered in terms of the Mining Law of 1872 and coal mines by the National Environmental Programme. The Mining Law of 1872 had its basis in encouraging mining on public land. This law is currently being revised and ultimately should compare with the regulations for coal mining.

For new hard rock mines, Figure 3 presents a schematic flow chart of the process to obtain mining approval.

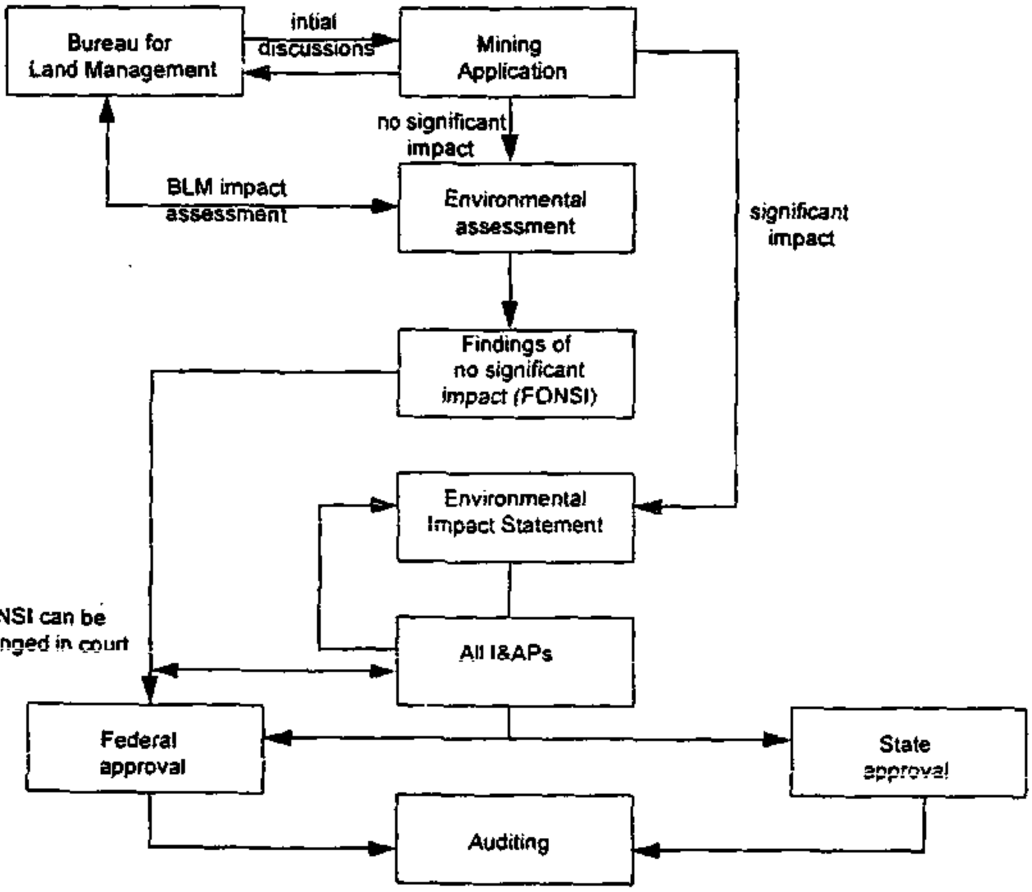

Figure 3 : Schematic Representation of Application for Mining

The current situation is that should initial discussions with the Bureau for Land Management indicate that there will be no significant environmental impact, then an environmental assessment is prepared. Usually this is done by the BLM themselves. Once this study indicates no significant impact, a finding of no significant impact (FONSI) will be prepared. This, however, is not the final situation, as certain FONSI's have been challenged successfully in court.

The preferred option, and the way all new mining applications will probably go, is for a detailed environmental impact statement to be produced. This is also done by the BLM or its consultants, but the mine will pay the costs. Costs range from $1 to $2 million for a single EIS.

The EIS is assessed rigorously by all Interested and Affected Parties until the Federal authorisation is given. The mine must also comply with State regulations which may address specific environmental issues for that particular region.

Auditing of the mine's environmental programme is done regularly. Every 3 months the BLM will audit the reclamation and mining operations. Air pollution is audited every 6 months. In terms of the Water Pollution Control Permit, the mine submits water quality reports monthly and are visited every six months for an on-site audit.

6. LEGAL & REGULATORY FRAMEWORK IN CANADA

6.1 WATER POLLUTION REGULATIONS

The province of Ontario is a major mining region. A number of different metals are mined, including gold, copper, nickel, lead, zinc, iron, silver and uranium. Gold accounts for approximately 60 percent of the total mining production.

The regulation of effluent from different industrial sectors is done by way of proclamation and enforcement of effluent monitoring and effluent limits for the various sectors. In the development of effluent limit regulations for the metal mining sector, a number of studies were undertaken.

In 1991, a detailed water quality monitoring programme was instituted to classify mine effluent streams. Each mine was sampled at a number of sites, at a frequency ranging from 12 per year to 150 samples per year. Each sample was analysed for 150 different compounds at a cost of approximately $18 million (Canadian). The results of the monitoring programme were published as a status report which sets out the quality of water being discharged from each mine.

In addition to this status report, a study of Best Available Pollution Control Technology was undertaken by a firm of consultants. This investigated and evaluated a world-wide scan of pollution control technologies, which would be applicable to Canadian and other mines with similar ores and climate. (This aspect of climate is distinctive as far as South Africa is concerned, in that there is an over-abundance of water in Ontario. This is not only from the rainfall but from the snow melt during spring - a situation not experienced in South Africa.) The results of this survey of technology were published as were a study on Risk Assessment and Mine Development and Decommissioning.

All these studies led to the development of draft regulations for effluent limits for the metal mining sector in late 1994. Released by the Ministry of Environment and Energy, they are stringent and apply to all metal mines discharging more than 50 m^3/day. From the status report on mine water effluent qualities, a total of 42 compounds were identified which routinely exceed the guidelines. Of these, 7 were regulated in the effluent limit regulations. These are total cyanide, total suspended solids, lead, copper, nickel, zinc and arsenic. Additionally, the pH must be between 6 and 9,5. As a catch-all to ensure that none of the other potentially toxic compounds are within the effluent, trout must be able to survive in undiluted mine effluent for 96 hours, and daphnia must survive for 48 hours.

The effluent limit regulations apply to all metal producing mining plants, both existing and new plants. The following sections are contained within the regulations:

i. **Application**

This defines the type of operations covered, as well as the conditions under which the regulations continue to apply. This ensures that the regulations apply when the mine temporarily closes for shutdown, strikes, and when the mine reduces its discharge to below 50 m^3/day. The moment the mine discharges more than 50 m^3/day the regulations come into force and remain so until closure.

ii. **Production**

The common use of meanings of production are the basis for the definition. Any production, no matter how small or large, nor for what purpose is included.

iii. **Metal Mining Plant**

This includes all surface and underground workings, buildings, roads, infrastructure, all waste disposal sites, waste water treatment sites, pollution control measures, and any refinery, metallurgical plant, furnace etc.

iv. **Effluent types.**

The following effluent types are defined:

<u>Process effluent</u> - This is defined as effluent that has, by design, come into contact with process materials (this includes waste rock, tailings, slag etc.), blowdown water, effluent that results from cleaning or maintenance during shutdown, and any of these combined with cooling water effluent or stormwater effluent.

<u>Cooling water effluent</u> - This refers to water that is used within the processes to remove heat but which has not come into contact with the process materials.

Overflow effluent - This is defined as effluent that is discharged from a tailings area through a spillway which has been designed to prevent collapse as a result of an extraordinary thaw or storm event. It must be noted, that in terms of the regulation, discharge from a tailings dam can only occur as a result of a weather-related event.

Stormwater effluent - This refers to run-off from a storm event or thaw that is not used in any industrial process.

v. Monitoring Requirements

Process effluent requirements.

As mentioned above, all process effluents must meet the following requirements:

* pH between 6 and 9.5
* Non-toxic to rainbow trout and *Daphnia magna*.

Sampling must be carried out three times per week for pH, total suspended solids and total cyanide. Sampling once a week is required for lead, copper, nickel, zinc and arsenic.

The toxicity tests for daphnia and trout must be carried out monthly. If twelve successive tests are passed, then this can be reduced to quarterly.

Table 2 : Applicable water quality concentrations

Parameter	Monthly Average Concentration mg/ℓ	Daily Concentration mg/ℓ
Total Cyanide	1	2
Total suspended solids	15	30
Lead	0,2	0,4
Copper	0,3	0,6
Nickel	0,5	1
Zinc	0,5	1
Arsenic	0,5	1

Cooling water effluent - Weekly samples are required for suspended solids, the effluent must be non-toxic for trout and daphnia (tests to be done monthly). The frequency can change to quarterly if twelve successive samples are passed.

Overflow effluents - Assessment monitoring, once every 8 hours, for total cyanide, total suspended solids, copper, lead, nickel, zinc and arsenic.

Stormwater effluents - A special monitoring protocol for stormwater monitoring has been drawn up by the Ministry and is available at the mine.

vi. **Flow measurement**
The volume of process streams must be measured to an accuracy of 15 percent and cooling and overflow effluents at 20 percent.

vii. **Types of samples**
Composite samples are generally required.

viii. **Sampling points**
A sampling point must be established on each of the effluent water circuits. This site is then registered with the Ministry.

iv. **Calculation of loads**
The regulations do not place limits on loads. However, the loads must be calculated for reporting for each of the effluent streams.

x. **Calculation of concentrations**
The regulations place limits on the concentrator.

xi. **Cyanide**
If cyanide is not used within the plant, then analyses need not be performed.

The results of each of these analyses are recorded and kept on the mine for a period of three years. Summary quarterly reports are sent to the Ministry, but more frequent reports can be requested.

The enforcement of the regulations is strict. For the first offence a fine may be imposed of up to $50 000 or $50 000 per day until the problem is rectified. The second offence warrants a fine of $100 000. If the mine is caught deliberately manipulating the regulations, the Director or Managing Director of the mine may be jailed.

6.2 MINING ENVIRONMENTAL REGULATIONS.

Until recently, there were no regulations pertaining to a mine's environmental impact. This has resulted in some mines having no plan of action for closure of the mine. An example is Falconbridge mines. Tailings are discharged into a natural valley in the Moose Lake system. Those tailings with a high pyrite content are placed at the lowest point of the valley and then overlain with 3 metres of normal tailings. The long term pollution problems are clearly evident but the mine has no indication of what must be done on closure. All water from the tailings, the natural runoff and snow melt are dammed by Moose Lake. The outflow from Moose Lake is treated with lime and the water discharged to a settling dam. Outflow from the settling dam is treated with carbon dioxide to lower the pH. This water is then released to the river. The total cost is some $ 500 000 per annum.

Recently, the Ministry of Environment and Energy implemented a more rigid strategy for new mines. The process is presented in Figure 4.

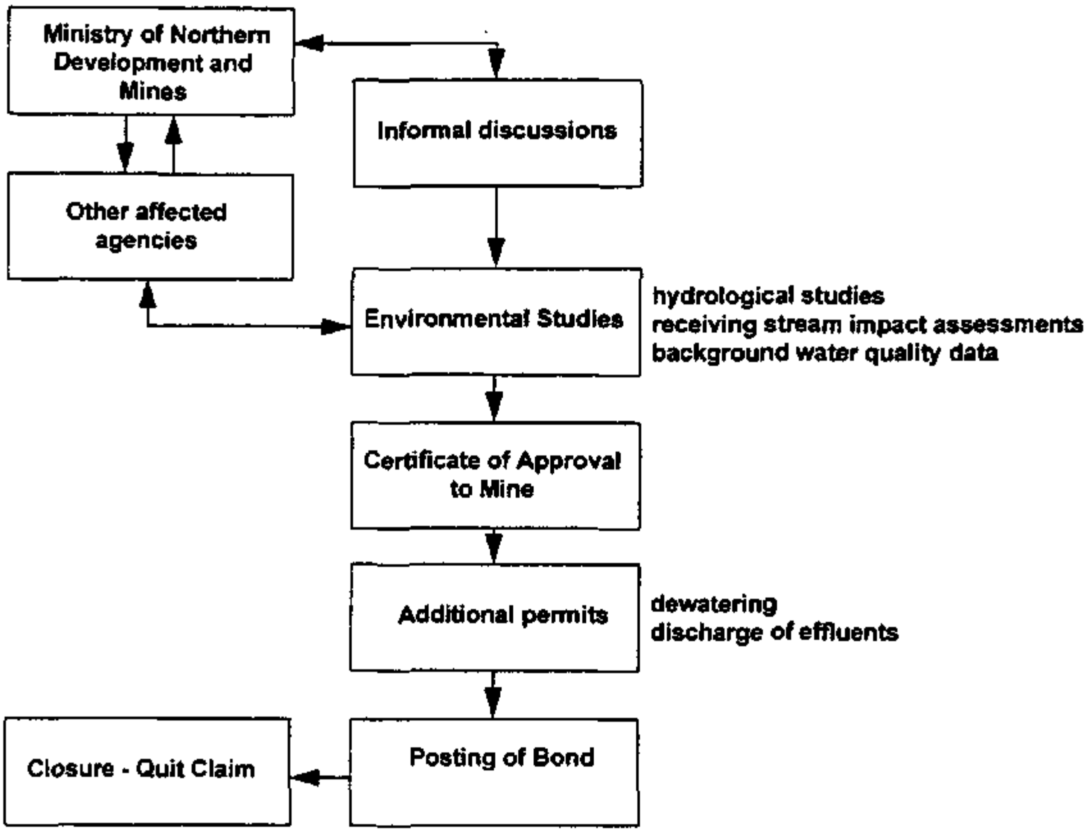

Figure 4 : Mining Application Process

The province of Ontario requires the mines to post a bond and also to supply detailed plans for closure and details of a monitoring programme. The existing situation for closure, is that the authorities sign a quit claim which then allows the mine to walk away from the site. No quit claims have been signed. The proposed procedure is to develop an exit ticket which will only be approved once all the involved agencies are satisfied with the closure plans. Additional information is given in the Overseas Study Tour Report.

7. LEGAL & REGULATORY FRAMEWORK IN THE UNITED KINGDOM

The National Rivers Authority (NRA) is a major environmental protection agency with the task of safeguarding and improving the natural water environment in the United Kingdom.

Established in 1989, the NRA is an independent public body. It is accountable to Parliament and has statutory responsibilities and powers in relation to a number of important functions including pollution control, water resources, flood defence, fisheries, conservation and recreation.

Policy issues are coordinated by the NRA's Head Office in Bristol and policy decisions are made by a national board consisting of 15 men and women appointed by the government (Department of the Environment and the Ministry of Agriculture, Fisheries and Food). Authorities are subdivided into actual river catchments. A number of committees advise these authorities on their work. These include statutory rivers advisory committees, flood defence committees and fisheries advisory committees. Meetings are open to the press and public and an active pollution prevention philosophy underlies all water quality work.

Groundwater and quantity of surface flow are addressed as major concerns by the NRA. The NRA is also responsible for a flood warning service through weather and river level monitoring by remote sensing(9).

Recent events have highlighted both the potential and actual environmental damage resulting from abandoned mines and the legal and practical difficulties which the NRA faces in its efforts to prevent and clean-up pollution arising from such mines. Although closely associated with the nature and problems of contaminated land, abandoned mines have sufficient unique characteristics to merit an evaluation in their own right.

The NRA's interests are therefore as follows:

- In the first instance it is necessary to evaluate the extent to which existing discharges from mines already abandoned, affect water quality and would therefore need to be tackled if water quality is to be improved.

- Secondly, there is a need to ensure that the continuing abandonment of existing mines results in the minimal amount of environmental impact, and that it is carried out within the limited protection of the existing law.

- Thirdly, in order to maintain the existing quality of waters, and to ensure that the effort currently expended to improve water quality is not hindered, particularly if such improvement programmes are related to meeting water quality objectives which are to be set on a statutory basis - it is important to ensure that mining activities in the future are properly controlled.

In the NRA's short existence it has already given evidence to the following:

- Welsh Affairs Committee on the Future of Open cast Coal mining (March 1991);

- Welsh Affairs Committee on Water Pollution from Abandoned Mines (March 1992);

- Department of Trade and Industry on the Privatisation of British Coal (June 1992); and

- Department of Trade and Industry on its Review on the Pits Proposed for Closure by British Coal (November 1992)

In its evidence to all of these the NRA has repeatedly called for changes in the legal position of mine abandonment, and has said that the entire cycle of mine planning, opening, closure and possible reopening needs to he re-examined. These concerns were echoed in the sixteenth report of the Royal Commission on Environmental pollution published in June 1992, and the Government has since stated in "This Common Inheritance - The Second Year Report" (October 1992) that it is "...considering the framework of legal responsibility for pollution in abandoned mines".

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WATER QUALITY STANDARDS

1. INTRODUCTION

Water quality standards, criteria and guidelines define the limits for the use of water in various processes. Processes in this context refer to not only industrial processes such as gold extraction, but also to processes such as agricultural production, potable supply of water, and ecological processes.

At the outset, some clarification of terms is required. Generally, a criterion is the concentration of a contaminant above which the water cannot be used for its intended purpose(2). Since the use of water for any purpose is seldom restricted to a single parameter, the individual criterion for a number of potential contaminants make up the criteria for a particular use. An example is the drinking water criteria for the supply of potable water.

Criteria are not regulatory requirements but merely serve as guidelines, upon which a regulatory authority may base water quality standards(2). Standards, on the other hand, are regulatory requirements and as such are legislatively enforceable. Whereas criteria can be expected to be based on the optimum quality of water for a particular use, standards are influenced by political and economic factors, and as such may represent a compromise.

In the mining context, both criteria and standards apply to various processes. The water quality needs for specific processes have specified criteria which set the boundaries for the use of a particular water. Water discharged from mines must conform to at least the general effluent standard. At present the Department of Water Affairs is changing the emphasis of pollution control away from a uniform standard towards a receiving water quality objectives approach. This aspect will be discussed in detail later in this review. Once water enters the "public stream" other users have specific needs in terms of water quality. The influence, therefore, of the mine on downstream water quality is an important factor which requires assessment against specific fitness for use criteria.

2. MINE WATER QUALITY GUIDELINES

Water quality guidelines have been set for a number of processes within normal mining operations. These are for :

- mine service water
- hydropower water
- cooling water

The guidelines are presented in Tables 1 to 3(4).

Table 1 : COMRO Guidelines For Mine Service Water

Parameter	Unit	Range and Upper Limits[a]
pH[b]	pH unit	5,5 - 8,5
Langelier saturation index	-	Non-scaling
$CaSO_4$ Precipitation Potential	mg/ℓ	< 0
Suspended solids[c]	mg/ℓ	30
Total Chlorine	mg/ℓ as Cl_2	0,5 - 1
Total Dissolved Solids	mg/ℓ	10 000
Chloride[d]	mg/ℓ as Cl	Limited by TDS

[a] Range and upper limits specified for carbon steels

[b] pH range for galvanised steels: 7,0 - 8,5

[c] Suspended solids for close tolerance equipment, e.g. hydropower systems: 5 mg/ℓ

[d] Chloride: for chromium steels determined by critical chloride concentration - 250 mg/ℓ

Table 2 : COMRO Guidelines For Hydro-Power Water

Parameter	Unit	Guideline values		
		Carbon Steel	Chromium Steel	Galvanised Steel
pH	pH unit	6 - 8,5	6 - 8,5	7 - 8,5
Langelier Saturation index	-	0,5 to 0 at 35 °C		
Product of calcium and sulphate ion concentrations	-	Not to exceed apparent CaSO4 solubility product at 0 °C for the particular water composition		
Particle size distribution	-	To ISO 4406 class 19/15 or lower		
Suspended solids	mg/ℓ	5	5	5
Turbidity	NTU	3	3	3
Standard plate count	per 1 mℓ	< 100	< 100	< 100
Total dissolved solids (TDS)	mg/ℓ	< 1 500	Limited by Cl	Limited by Cl
Chloride	mg/ℓ	Limited by TDS	< 250[a]	< 3 000

[a] Not to exceed the critical chloride concentration of the particular water; should be taken as 250 mg/ℓ if not known

NTU = Nephelometric turbidity unit

These guidelines have been determined for the more susceptible processes within mining operations. Potable water criteria will be presented under the section addressing general water quality guidelines.

Table 3 : COMRO Guidelines For Condensor Cooling Water

Parameter	Unit	Range and Upper Limits				
		Stainless steel[b]	Copper	70/30 Cu- Ni	90/10 Cu- Ni	Titanium
pH[a]	pH unit	3,0-8,5	5,5-8,5	5,5-8,5	5,5-8,5	1,0-8,5
Suspended solids	mg/ℓ	30	30	30	30	< 80
Langelier Saturation Index[a]	-	ns	ns	ns	ns	ns
CaSO4 Precipitation Potential[a]	mg/ℓ	ns	ns	ns	ns	ns
Free Chlorine	mg/ℓ as Cl_2	10	2	4	4	No limit
Ammonia	mg/ℓ as NH_4^+	No limit	400	200	2 000	No limit
Ferrous iron	mg/ℓ as Fe^{2+}	40	10	10	40	40
Total Dissolved Solids	mg/ℓ	10 000	10 000	10 000	10 000	10 000
Chloride	mg/ℓ	250	Lbt	Lbt	Lbt	Lbt

[a] The "non-scaling" properties of mine water can be adjusted by the use of proprietary brand conditioning chemicals which will determine the upper limit for the pH, Langelier Saturation Index and the $CaSO_4$ Precipitation Potential.

[b] AISI 316L or more highly alloyed practical estimate.

ns = non-scaling

Lbt = Limited by Total Dissolved Solids

3. EFFLUENT DISCHARGE STANDARDS

In South Africa, the Uniform Effluent Standards (UES) have been in place for over 20 years(12). The UES approach aims to regulate the input of effluents into the river systems by way of uniform standards. The net goal is to approach zero discharge of effluents. The standards are therefore set to treat contaminated effluents on the basis of BATNEEC (best available technology not entailing excessive cost).

This approach, however, has a number of drawbacks(12). The prime focus is on the effluent and not on the receiving water. This ignores the fact that there may be more than one discharge, or that there may be high background concentrations in the river system. Also, the ability of the river system to assimilate pollutants is not brought into the equation. On the plus side however, it is relatively simple, is comprehensible and is straightforward in application.

Although the application of the UES has ultimately led to a decrease in the rate of water quality deterioration(12), deterioration of the resource continued. It was realised that a different more advanced approach to water quality management was required. In 1990, the Department of Water Affairs & Forestry (DWA&F) adapted its approach to one where the following principles would be included:

" The desired quality of a water resource is determined by its present and/or intended uses. This quality should be stated as a list of water quality objectives.

It is accepted that the water environment has a certain, usually quantifiable, capacity to assimilate pollutants without detriment to predetermined quality objectives.

The assimilative capacity of a water body is part of the water resource and, as such, must be managed judiciously and shared in an equitable manner amongst all water users for the disposal of their wastes.

For those pollutants which pose the greatest threat to the environment, because of their toxicity, extent of bio-accumulation and persistence, a precautionary approach aimed at minimising or preventing inputs to the water environment should be adopted. "(12).

The Department's new approach includes both the principles of RWQO to non hazardous pollutants, and pollution prevention for hazardous substances.
It can therefore, be seen that the control of point source pollutants can be controlled via the use of regulations and permits. Monitoring of effluents therefore becomes an important factor. Monitoring programmes are used to gather information on water quality, how it changes over space and time and how it responds to management actions.

Having said this, most mines are still under the restrictions of the general effluent standards. Table 4 presents these standards[4, 7].

Table 4 : General Effluent Standards

Parameter	**Standard**
Colour, odour or taste	Shall not contain any substance in a concentration capable of producing any colour, odour or taste
pH	Shall be between 5,5 and 9,5
Dissolved oxygen	Shall be at least 75 per cent saturation
Typical (faecal) coliforms	Shall not contain any typical (faecal) coliforms per 100 millilitres
Temperature	Shall be a maximum of 35 °C
Chemical oxygen demand	Not to exceed 75 mg/ℓ after applying the chloride correction
Oxygen absorbed	The oxygen absorbed from acid N/80 $KMnO_4$ in 4 hours at 27 °C shall not exceed 10 mg/ℓ
Conductivity	Not to be increased by more than 75 mS/m (25 °C) above that of the intake water, with an absolute maximum for any effluent of 250 mS/m
Suspended solids	Not to exceed 25 mg/ℓ
Sodium	Not to be increased by more than 90 mg/l above that of the intake water
Soap, oil or grease	Not to exceed 2,5 mg/ℓ
Other Constituents:	**Maximum concentrations**
Residual chlorine	0,1 mg/ℓ
Free and saline ammonia	10,0 mg/ℓ as N
Arsenic	0,5 mg/ℓ as As
Boron	1,0 mg/ℓ as B
Hexavalent chromium	0,05 mg/ℓ as Cr
Total chromium[a]	0,5 mg/ℓ as Cr
Copper[a]	1,0 mg/ℓ as Cu
Phenolic compounds	0,1 mg/ℓ as phenol
Lead[a]	0,1 mg/ℓ as Pb
Cyanides	0,5 mg/ℓ as CN
Sulphides	1,0 mg/ℓ as S
Fluoride	1,0 mg/ℓ as F
Zinc	5,0 mg/ℓ as Zn
Manganese	0,4 mg/ℓ as Mn
Cadmium[a]	0,05 mg/ℓ as Cd
Mercury[a]	0,02 mg/ℓ as Hg
Selenium	0,05 mg/ℓ as Se
Other	The waste water or effluent shall contain no other constituents in concentrations which are poisonous or injurious to humans, animals, fish other than trout, or other forms of aquatic life, or which are deleterious to agricultural use

[a] The sum of the concentrations of Cadmium (as Cd), Chromium (as Cr), Copper (as Cu), Mercury (as Hg) and Lead (as Pb) shall not exceed 1 mg/ℓ.

On the international scene, restrictions have also been placed on mines, in terms of their discharge to surface water systems. This includes both active discharge (i.e. pumpage) or passive discharge (i.e. seepages).

For comparison, effluent quality standards for Canada, USA, and Europe are presented in Tables 5,6,7 and 8[1].

An examination of Tables 4 and 5 - 8 indicates that some differences exist between the South African general effluent standard and those applicable internationally.

Table 5 : U.S. Effluent Regulations - Copper, Lead, Zinc, Gold, Silver, and Molybdenum Ores

Parameter	(1)		(2)		(3)		(4)		(5)		(6)		(7)		(8)		(9)	
	max.	avg	max.	avg	max.	avg	max.	avg	max.	avg	max.	avg	max.	avg	max.	avg	max.	avg
total susp. solids (mg/ℓ)	30	20	30	20	30	20	30	20	30	20							30	20
pH	6.0 - 9.0		6.0 - 9.0		6.0 - 9.0		6.0 - 9.0		6.0 - 9.0		6.0 - 9.0						6.0 - 9.0	
Cd (mg/ℓ)			0.10	0.05	0.10	0.05			0.10	0.05	0.10	0.05	0.10	0.05	0.10	0.05	0.10	0.05
Cu (mg/ℓ)	0.30	0.15	0.30	0.15	0.30	0.15			0.30	0.15	0.30	0.15	0.30	0.15	0.30	0.15	0.30	0.15
Zn (mg/ℓ)	1.5	0.75	1.0	0.5	1.0	0.5			1.0	0.5	1.0	0.5	1.5	0.75	1.0	0.5	1.5	0.75
Pb (mg/ℓ)	0.6	0.3	0.6	0.3	0.6	0.3							0.6	0.3	0.6	0.3	0.6	0.3
As (mg/ℓ)					1.0	0.5			1.0	0.5	1.0	0.5						
Hg (mg/ℓ)	0.002	0.001	0.002	0.001									0.002	0.001	0.002	0.001	0.002	0.001

BEST PRACTICAL CONTROL TECHNOLOGY (BPT):

(1) concentration of pollutants discharged in mine drainage for copper, lead, zinc, gold and silver ores.

(2) concentration of pollutants discharged from mills which employ froth flotation for copper, lead, zinc, gold and silver ores.

(3) concentration of pollutants in mine drainage from mines producing 5,000 metric tons or more of molybdenum ores per year.

(4) concentration of pollutants in mine drainage from mines or mills producing less than 5,000 metric tons of molybdenum ores per year by methods other than ore leaching

(5) concentration of pollutants discharged from mills processing 5,000 tons or more of molybdenum ore per year by purely physical methods including ore crushing, washing, jigging, and heavy media separation

(6) concentration of pollutants discharged from mills processing 5,000 metric tons or more of molybdenum ores per year by froth flotation methods.

BEST AVAILABLE TECHNOLOGY ECONOMICALLY ACHIEVABLE (BAT):

(7) concentration of pollutants discharged in mine drainage for copper, lead, zinc, gold and silver ores

(8) concentration of pollutants discharged from mills which employ froth flotation for copper, lead, zinc, gold, silver, or molybdenum ores

NEW SOURCE PERFORMANCE STANDARDS - BEST AVAILABLE DEMONSTRATED TECHNOLOGY (BADT):

(9) concentration of pollutants discharged in mine drainage from mines producing copper, lead, zinc, gold, silver, or molybdenum ores.

Table 6 : U.S. Effluent Regulations - Iron, Uranium, and Nickel

Parameter	Iron		(1) Uranium		(2) Uranium		(6) Uranium		(7) Uranium		(3) Nickel		(4) Nickel		(5) Nickel	
	max.	avg	max.	avg	max.	avg	max.	avg	max.	avg	max.	avg	max.	avg	max.	avg
total suspended solids (mg/ℓ)	30	20	30	20	30	20			30	20	30	20	50	30	30	20
Fe, dissolved (mg/ℓ)	2.0	1.0														
pH	6.0 - 9.0		6.0 - 9.0		6.0 - 9.0				6.0 - 9.0		6.0 - 9.0		6.0 - 9.0		6.0 - 9.0	
Cd (mg/ℓ)											0.10	0.05			0.10	0.05
Cu (mg/ℓ)											0.3	0.15			0.30	0.15
Zn (mg/ℓ)			1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5	1.0	0.5			1.0	0.5
Pb (mg/ℓ)													0.6	0.3		
As (mg/ℓ)					1.0	0.5					1.0	0.5			1.0	0.5
Hg (mg/ℓ)																
COD (mg/ℓ)			200	100		500	200	100	200	100						
Ra-226, dissolved (pCi/ℓ)			10	3	10	3	10	3	10	3						
Ra-226, total (pCi/ℓ)			30	10	30	10	30	10	30	10						
U (mg/ℓ)			4	2			4	2	4	2						
NH_3						100										

BEST PRACTICAL CONTROL TECHNOLOGY (BPT):

(1) concentration of pollutants discharged in mine drainage for uranium, radium, and vanadium ores

(2) concentration of pollutants discharged from mills which employ the acid leach, alkaline leach or combined leach process for the extraction of uranium, radium, and vanadium ores

(3) concentration of pollutants discharged in mine drainage from mines producing 5000 metric tons or more of nickel ores per year

(4) concentration of pollutants discharged in mine drainage from mines producing less than 5000 metric tons of nickel ore per year by methods other than ore leaching

(5) A/ concentration of pollutants discharged from mills processing 5000 metric tons or more of nickel ore per year by purely physical methods including ore crushing, washing, jigging, heavy media separation and magnetic and electrostatic separation
B/ concentration of pollutants discharged from mills processing 5000 metric tons or more of nickel ore per year by froth flotation methods

BEST AVAILABLE TECHNOLOGY ECONOMICALLY ACHIEVABLE (BAT):

(6) concentration of pollutants discharged in mine drainage from mines that produce uranium, including plants using in-situ leach methods

NEW SOURCE PERFORMANCE STANDARDS (NSPS) - BEST AVAILABLE DEMONSTRATED TECHNOLOGY (BADT):

(7) concentration of pollutants in mine drainage from mines that produce uranium ore, excluding mines using in-situ leach methods.

3.1 UNITED STATES OF AMERICA

A comparison with the effluent standards applicable to gold ore systems shows that the general effluent standards are lower in respect of suspended solids, lead, and arsenic, but are higher for copper, zinc and mercury.

Table 7 : Canadian Effluent Limits

PARAMETER	ONTARIO **	QUEBEC	BRITISH COLUMBIA range ***	SASKATCHEWAN		NEWFOUND-LAND	CANADA *		
				mean	max.		monthly mean	comp sample	grab sample
Cd (mg/ℓ)	0,001		0,01 - 0,1			0,05			
Hg (mg/ℓ)	0,001		Nil - 0,005			0,005			
Suspended solids (mg/ℓ)	15	25,0	25 - 75			30	25,0	37,5	50,0
pH	5,5 - 10,8	6,5 - 9,5	6,5-8,5 6,5-10	6,0-9,5	5,0-10,0	5,5 - 9,0	>6,0	>5,5	>5,0
As, total (mg/ℓ)	0.5	0,50	0,10 - 1,0	0,5	1,0	0,5	0,5	0,75	1,0
As, trivalent (mg/ℓ)			0,05 - 0,25						
Cu (mg/ℓ)	1,0 (0,3)	0,30	0,05 - 0,03	0,3	0,6	0,3	0,3	0,45	0,6
Ni (mg/ℓ)	1,0 (0,5)	0,50	0,2 - 1,0	0,5	1,0	0,5	0,5	0,75	1,0
Pb (mg/ℓ)	1,0 (0,2)	0,20	0,05 - 0,2	0,2	0,4	0,2	0,2	0,3	0,4
Zn (mg/ℓ)	1,0 (0,5)	0,50	0,2 - 1,0	0,5	1,0		0,5	0,75	1,0
Fe (mg/ℓ)	1,0	3,00	0,3 - 1,0						
Total CN (mg/ℓ)	2,0 (1,0)	1,50	0,1 - 0,5	1,0	2,0	0,025			
Phenols (mg/ℓ)	0,02								
BOD$_5$ (mg/ℓ)	15								
Phosphorus (mg/ℓ)	1,0								
Gross Alpha pCi/ℓ			10 - 100						
Radium pCi/ℓ			less than 10						
Al (mg/ℓ)			0,5 - 1,0						
Ammonia as N (mg/ℓ)	10		1,0 - 10,0						
Sb (mg/ℓ)			0,25 - 1,0						
Cr (mg/ℓ)			0,05 - 0,3						
Co (mg/ℓ)			0,5 - 1,0						
F (mg/ℓ)			2,5 - 10,0						
Mn (mg/ℓ)			0,1 - 1,0						
Mo (mg/ℓ)			0,5 - 5,0						
Nitrite/Nitrate as N (mg/ℓ)			10,0 - 25,0						
Phosphate as P (mg/ℓ)			2,0 - 10,0						
Se (mg/ℓ)			0,05 - 0,5						
Ag (mg/ℓ)			0,05 - 0,5						
Free CN (mg/ℓ)		0,10							
Dissolved Solids			2500 - 5000			1000			
UO-			2,0 - 5,0						
Oil and Grease	15,0		10,0 - 15,0						
Uranium				2,5	5,0				
Radium-226 (Bq/ℓ)				0,37	1,11	0,37			
Thorium-230 (Bq/ℓ)				1,85	3,7				
Lead-210 (Bq/ℓ)				0,92	1,84				
Un-Ionized Ammonia				0,5	1,0				
Temperature						>32°C			
Strontium 90						0,37			
oils (ether extract)						15			
Ba						5,0			
Bo						5,0			
Cl						1,0			
Cr- trivalent						1,0			
Cr- hexavalent						0,05			
toxicity			80% - 100%						

*** Federal guidelines used by Manitoba, Nova Scotia and New Brunswick.**

**** Typical values more recently applied on Certificates of Approval (OWRA Section 24) in Ontario.**

***** Lower limits are for new plants, higher limits are for older plants, limits are guidelines as opposed to regulations.**

3.2 CANADA

For the purpose of comparison, the British Columbia standards have been used. The general effluent standards are lower for suspended solids, arsenic, lead, fluoride, manganese, selenium, total dissolved solids and oil and grease. Variables which have general effluent standards higher than the British Columbia standard are mercury, copper and zinc, and ammonia and cyanide are the same.

Table 8 : International Effluent Limits

PARAMETER (mg/ℓ)	CANADA			GERMANY (federal)	GERMANY (regional)	SWEDEN	FINLAND[2] (operating approval limits)	ITALY	SPAIN[3]			TASMANIA (Australia) (4)	
	monthly mean	comp sample	grab sample						(1)	(2)	(3)	Inland	Marine
Cd				0.2	0.2			0.02	0.5	0.2	0.1	0.01	0.01
Hg				0.05	0.05			0.005				0.002	0.002
Suspended solids	25.0	37.5	50.0			15	10 - 60	80	300	150	80		
pH	>6.0	>5.5	>5.0				5.5 - 9.5	5.5 - 9.5	5.5 - 9.5				
Pb	0.2	0.3	0.4	0.5	0.5	0.1		0.2	0.5	0.2	0.2	0.05	0.2
Zn	0.5	0.75	1.0	1	1	0.1	1.0	0.5	20	10	3		
As	0.5	0.75	1.0	0.1	0.1			0.5	1.0	0.5	0.5	0.05	0.25
Cu	0.3	0.45	0.6	0.5	0.5	0.1	0.05 - 0.3	0.1	10	0.5	0.2	1.0	1.0
Cr.hexavelant				0.5/0.1	0.1			0.2				0.05	0.05
Ni	0.5	0.75	1.0		0.5	0.1	0.3 - 1.0	2	10	3	2		
Mineral Oil						0.5		5					
Ra-226 (pCi/ℓ)	10.0	20.0	30.0										
Oil & Grease												10	
CN					0.1		0.5	0.5				0.05	0.2
Ammonia-N									50	50	15	0.5	0.5
Sulphide				1	1								
Sulphate							1200 - 1600		2000	2000	2000		
Nitrate									1.0	0.5	0.5		
Al								2	2	1	1		
Cr (Total)									4	3	2		
Fe							2 - 3	2	10	3	2		

(1) Guidelines - each operation is assessed on a case by case basis
(2) range of discharge limits in effect at six base metal mines.
(3) See section 4.4 - Spain, for differentiation between limits (1), (2), and (3).
(4) discharge limits to inland waters, discharge limits to marine waters

3.3 GERMANY

For the purpose of comparison, the federal limits have been used. The general effluent standard is lower for cadmium, mercury and lead, and are higher for zinc, arsenic, copper and cyanide. The standard for sulphide is the same.

3.4 SWEDEN

The general effluent standards are higher than the Swedish standard for suspended solids, zinc and copper and are the same for lead.

3.5 ITALY

The general effluent standards are lower than the Italian standard for suspended solids and lead but are higher for cadmium, mercury, zinc and copper. There is no difference for pH range and arsenic.

3.6 TASMANIA (AUSTRALIA)

The general effluent standards are higher than the Tasmanian standards for cadmium, mercury, lead, arsenic, copper and cyanide. The general effluent standard is not lower for any of the mutual compounds.

3.7 SUMMARY OF COMPARISONS

In general, the South African general effluent standards are higher than international standards for copper, mercury and zinc, lower for lead and suspended solids, and similar for arsenic, cyanide and cadmium. As identified above, there are other specific differences between individual international standards.

3.8 RADIOACTIVITY

There are at present, no limits set for radioactivity levels within General Effluent Standards for the disposal of effluents into public streams nor in the SABS specification for drinking water(7). The SABS recommends compliance with the International Commission on Radiological Protection (ICRP) limits. The following accepted standards are applicable:

^{226}Ra: 0,19 Bq/ℓ
Uranium: 44 µg/ℓ

In 1987, recommendations by the ICRP, reaffirmed the limit on annual dose equivalent to whole body for workers of 50 mSv introduced in 1956. A detailed review of radioactivity and its impact on various users of water has been prepared as a separate literature review.

4. GENERAL WATER QUALITY GUIDELINES

4.1 INTRODUCTION

"Water quality guidelines serve as the scientific basis for the understanding and the quantification of user water quality requirements as well as a tool to assess the impact of physical development on the users"(8). These users have been identified as :

- agriculture
- industry
- domestic
- recreation
- the natural environment.

Water which may be fit for one user, may not necessarily be fit for another. This means that should the quality of water deteriorate and one user is affected, another user may not be. The net result is that the quality of water must be linked to a particular user(8). In determining this fitness for use, a baseline set of criteria as to the user's water quality needs are required. This allows an assessment to be made concerning the introduction of an increased level of a particular chemical. These criteria have been formulated by the Department of Water Affairs & Forestry into a series of water quality guidelines for agricultural, industrial, domestic and recreational users. As yet, there are no guidelines for the natural environment. There are however a number of other sources of criteria which can be used for the assessment of the fitness for use of a specific water for the natural environment .

These guidelines by DWA&F serve as a useful starting point in assessing the quality of water for identified users. With the advent however of the catchment approach to water management, specific catchments may have different guidelines depending on the activities within the catchment.

The water quality guidelines in the DWA&F series have defined ranges of values for various water quality parameters. Associated with each of the range of values is a description of the effect these have on the intended use of the water. This allows the water manager to determine the potential impact a particular quality of water will have on the user.

4.2 AGRICULTURE

For agriculture, there are three classes : Irrigation, livestock watering and aquaculture. The DWA&F have developed guidelines for each of these classes. A summary of the water quality guidelines is presented in Tables 9,10 and 11(11).

The guidelines for irrigation are centred around the potential impact water will have on certain crop species, and the influence of the sodium absorption ratio (SAR) on soil permeability. Increasing sodium salinity increases the SAR and this results in the formation of barriers within the soil profile, which impedes the movement of water to the root zone. Good farming practice can be used to manage this problem but the long term use of water with a high sodium salinity can lead to severe soil condition problems.

From the chemical point of view, boron is one of the main elements of concern. Although an essential plant nutrient, it becomes toxic at higher concentrations.

The DWA&F guidelines for agriculture class the water according to the following definitions:

Class 1 : The water can be used for even the most sensitive crops and soils without any reduction in yield or the need for special management practices.

Class 2 : The water can be used for all but the most sensitive crops and soils, with no reduction in yield or the need for special management practices.

Class 3 : Some yield loss is experienced even though special management practices are implemented, but a reasonable profit is realised.

Class 4 : Yield losses and/or the need for special management practices are such that the economic viability of irrigation is questionable. Certain crop types can, however, still be produced in special circumstances or by using special management practices.

For livestock watering, the palatability of the water is the main focus of the guidelines. However, the water requirements of the animals is also influenced by a number of physiological factors and the environment. These include conditions such as stage of physiological development, gestation, lactation, physical exertion, type of ration and dry matter intake, inorganic salt intake and ambient temperature - humidity indices.

Aspects which may affect the palatability of the water are algae and/or protozoa, hydrocarbons, pesticides and salts such as nitrates, sulphates, fluoride and the salts of the heavy metals.

Table 9 : DWA&F Guidelines For Irrigation

Water Quality Constituent	**Class I**	**Class II**	**Class III**	**Class IV**
Salinity (EC), mS/m	0-40	40-90	90-270	270-540
Sodicity (SAR), mmol/ℓ	0-1,5	1,5-3,0	3,0-5,0	5,0-10,0
Boron	0-0,2	0,2-00,9	0,9-1,5	1,5-3,0
Chloride	0-105	105-140	140-350	>350
Sodium (SAR, mmol/ℓ	0-3	3-5	5-7	7-9
Aluminium	0-5,0	0-5,0	5,0-10,0	10,0-20,0
Arsenic	0-0,1	0-0,1	0,1-1,0	1,0-2,0
Beryllium	0-0,1	0-0,1	0,1-0,25	0,25-0,5
Cadmium	0-0,01	0-0,01	0,01-0,025	0,025-0,05
Cobalt	0-0,05	0-0,05	0,05-2,5	2,5-5,0
Copper	0-0,2	0-0,2	0,2-2,5	2,5-5,0
Fluoride	0-2,0	0-2,0	2,0-7,5	7,5-15,00
Iron	0-5,0	0-5,0	5,0-10,0	10,0-20,0
Lead	0-0,2	0-0,2	0,2-1,0	1,0-2,0
Lithium	0-2,5	0-2,5	-	-
Manganese	0-0,2	0-0,2	0,2-5,0	5,0-10,0
Mercury	*No levels at present*			
Molybdenum	0-0,01	0-0,01	0,01-0,025	0,025-0,05
Nickel	0-0,2	0-0,2	0,2-1,0	1,0-2,0
Selenium	0-0,02	0-0,02	0,02-0,025	0,025-0,05
Tin	no levels at present			
Titanium	no levels at present			
Tungsten	no levels at present			
Uranium	0-0,01	0-0,01	0,01-0,05	0,05-0,1
Vanadium	0-0,1	0-0,1	0,01-0,05	0,05-1,0
Zinc	0-1,0	0-1,0	1,0-2,5	2,5-5,0
Nitrogen	0-5	5-30	>30	
pH	6,5-8,4			

All values are mg/ℓ except pH = pH units

For livestock watering, the palatability of the water is the main focus of the guidelines. However, the water requirements of the animals is also influenced by a number of physiological factors and the environment. These include conditions such as stage of physiological development, gestation, lactation,

physical exertion, type of ration and dry matter intake, inorganic salt intake and ambient temperature - humidity indices.

Aspects which may affect the palatability of the water are algae and/or protozoa, hydrocarbons, pesticides and salts such as nitrates, sulphates, fluoride and the salts of the heavy metals.

The following constituents are potentially hazardous and have a high incidence of occurrence in the natural aquatic environment:

salinity, chloride, sulphate, arsenic, copper, sodium, calcium, fluoride, molybdenum, magnesium, nitrates and nitrite and toxic algae.
The following constituents are potentially hazardous but have a low incidence of occurrence in the natural aquatic environment:

cadmium, chromium, mercury, lead, zinc, selenium, boron, aluminium, cobalt, iron, nickel, vanadium, manganese, radionuclides, pesticides and pathogens.

Table 10 : DWA&F Guidelines For Livestock Watering

Water Quality Constituents	Target Guideline Range
Arsenic	0-0,5
Calcium	0-1000
Chloride	0-1500 non-ruminants; 0-3000 ruminants
Copper	0-0,5 sheep; 0-1,0 pigs and poultry; 0-5 cattle
Fluoride	0-2 non-ruminants and dairy cattle; 0-6 ruminants
Molybdenum	0-0,01
Magnesium	0-500
Nitrate	0-100
Nitrite	0-10
Sodium	0-2000
Sulphate	0-1000
Aluminium	0-5
Boron	0-5
Cadmium	0-0,01
Chromium	0-1
Cobalt	0-1
Iron	0-10
Lead	0-0,1
Manganese	0-10
Mercury	0-0,002
Nickel	0-1
Pathogens	1000/100ml faecal coliforms; 5000/100ml total bacteria
Selenium	0-0,05
Vanadium	0-0,1
Zinc	0-20

All values are mg/ℓ except pH = pH units

In its broad sense, aquaculture is the cultivation and harvesting of aquatic organisms for commercial utilisation. In South Africa, freshwater aquaculture includes:

- the production of food;
- ornamental fish and plants for aquariums;
- recreational fishing;

- crocodile skins.

In South Africa, the development of guidelines for aquaculture are based on freshwater fish species. This approach was followed because freshwater fish production is presently the most important freshwater aquaculture activity in South Africa.

Table 11 : DWA&F Guidelines For Aquaculture

Constituent / properties	**Target guideline range**
Alkalinity	20-175 mg/ℓ $CaCO_3$
Ammonia	0-0,025 mg/ℓ unionised ammonia
Dissolved Oxygen	>5 mg/ℓ warmwater species
	>8 mg/ℓ cold water species
Carbon dioxide	2-5 mg/ℓ
Hydrogen sulphide	0-0,002 mg/ℓ
Nitrite	0-0,06 mg/ℓ
pH	6-9 pH units
Suspended solids	25-80 mg/ℓ
Total hardness	20-175 mg/ℓ $CaCO_3$

4.3 INDUSTRIAL

For industrial water quality guidelines, the emphasis is placed on the various users of water within the industrial sector, such as pulp and paper(10). There are no specific guidelines for mining. Criteria for water use within the mining industry have been presented earlier.

4.4 DOMESTIC

It is perhaps important to note that for South Africa, there are no standards for potable water quality. There are, however, guidelines, and at present those in most common user are the South African Bureau of Standards No 241 specification. These are presented in Table 12(6, 8).

Table 12 : SABS Specification For Drinking Water

Determinant	Unit	Recommended Maximum Limit	**Maximum Allowable Limit**
Colour	mg/ℓ Pt	20	NS
Odour and taste	-	Shall not be objectionable	Shall not be objectionable
Turbidity	NTU	1	5
pH	pH unit	6 - 9	5,5 - 9,5
Conductivity	mS/m	70	300
Macro-determinands	**mg/ℓ**		
Total hardness	$CaCO_3$	20 - 300	650
Magnesium	Mg	70	100
Sodium	Na	100	400
Chloride	Cl	250	600
Sulphate	SO4	200	600
Nitrate and Nitrite	N	6	10
Fluoride	F	1	1,5
Zinc	Zn	1	5

ICRP = International Commission for Radiological Protection
NS = Not specified
NTU = Nephelometric turbidity units

Table 12 : SABS Specification For Drinking Water - continued

Determinant	**Unit**	**Recommended Maximum Limit**	**Maximum Allowable Limit**
Micro-determinands	µg/ℓ		
Arsenic	As	100	300
Cadmium	Cd	10	20
Copper	Cu	500	1 000
Cyanide	CN	200	300
Iron	Fe	100	1 000
Lead	Pb	50	100
Manganese	Mn	50	1 000
Mercury	Hg	5	10
Phenolic compounds	Phenol	5	10
Selenium	Se	20	50
Bacteriological requirements			
Standard plate count	per 1 mℓ	100	NS
Total coliforms	per 100 mℓ	0	5
Faecal coliforms	per 100 mℓ	0	0
Radioactivity	If present, shall be within the limits laid down by the ICRP		

ICRP = International Commission for Radiological Protection
NS = Not specified
NTU = Nephelometric turbidity units

For comparison, the international standards for a number of countries and organisations are presented in Table 13[3].

Table 13 : Summary Comparison Of US Secondary Regulations With Canadian, EEC, And WHO Guidelines

Substance	**US Secondary Maximum Contaminant Level***	**Canadian Maximum Acceptable Limit***	**EEC**		**WHO Guideline Value**
			Guide Level *	**Maximum Admissible Concentration**	
Chloride	250mg/ℓ	250mg/ℓ	25 mg/ℓ	NS†	250 mg/ℓ
Color	15 cu	15 cu	1mg Pt-Co/ℓ	20 mg Pt-Co/ℓ	15 cu
Copper	1 mg/ℓ	1.0 mg/ℓ	100µg at treatment plant; 3.000 µg after 12 hours in piping	NS	1.0 mg/ℓ
Corrosivity	noncorrosive				
Fluoride	2 mg/ℓ	1.5 mg/ℓ		Water should not be aggressive Varies according to average temp *in the area*	1.5 mg/ℓ
Foaming agents	0.5 mg/ℓ	NS		NS	NS
Iron	0.3 mg/ℓ	0.3 mg/ℓ	50 µg/ℓ	300 µg/ℓ	0.3 mg/ℓ
Manganese	0.05 mg/ℓ	*0.05 mg/ℓ*	20 µg/ℓ	50 µg/ℓ	0.1 mg/ℓ
Odor	3 TON		0 dilution number	2 dilution number at 54°F (12°C)	
pH	6.5-8.5	6.5-8.5	6.5-8.5	NS	6.5-8.5
Sulfate	250 mg/ℓ	500 mg/ℓ	25 mg/ℓ	NS	400 mg/ℓ
Total dissolved solids	500 mg/ℓ	500 mg/ℓ	NS	NS	1000 mg/ℓ
Zinc	5 mg/ℓ	5 mg/ℓ	100µg at treatment plant; 5.000 µg after 12 hours in piping	NS	5.0 mg/ℓ
*Non enforceable †No standard					

A comparison between the SABS 241 and international potable guidelines indicates the following:

Generally, the SABS 241 maximum allowable limit exceeds all the international maximum levels. Only copper and fluoride are similar.

4.5 RECREATION

Aesthetic and health concerns tend to be the focus of the recreational use of water. This means that aspects such as litter, smell, and visual aspects are used to assess the fitness of use of the water for recreational purposes. The DWA&F guidelines are presented in Table 14(9).

Table 14 : DWA&F Guidelines For Recreation

Water Quality Constituent	Target guideline range
Algae	Contact recreation : free floating algae: 0-15 ug/ℓ attached filamentous algae should be absent Non-contact recreation : *free floating algae: 0-20 ug/ℓ*
Bilharzia	no numerical guideline proposed
Chemical irritants	no numerical guideline proposed
Clarity	Full contact / *intermediate contact recreation* 0-2,75 Secchi disk depth (m)
Coliphages	Full contact recreation : coliphages 0-20 counts /mℓ *non-contact recreation : not relevant*
Colour	no numerical guidelines proposed
Enteric viruses	Full /intermediate contact recreation 0-1 (TCID50 /10ℓ) non-contact recreation : not relevant
Faecal coliforms	Full contact recreation : F. coliforms : 0-150 counts/100mℓ E.coli. : 0-126 counts/100mℓ Intermediate contact recreation : *F. coliforms: 0-1000 counts/100 mℓ* E. coli : 0-1000 counts/100 mℓ non-contact recreation : not relevant
Floating matter	no numerical guideline proposed
Nuisance plants	no numerical guideline proposed
Odour	Recreational water should be free of any substances which cause noticeably unpleasant or objectionable odours
pH	Full / intermediate contact recreation : pH 6.5-8,5 Non-contact recreation : not relevant
Protozoan parasites	Full contact/ intermediate contact recreation < 1 Guardia cyst/10ℓ and < 1 Cryptosporidium oocyst /10ℓ *non-contact recreation : not relevant*

4.6 NATURAL ENVIRONMENT

Although no guidelines are presented as yet by the DWA&F, Kempster et al produced a set of criteria in 1980(5) which are generally accepted as being appropriate for the natural environment. Table 15 presents these criteria(5).

Table 15 : Water Quality Criteria For Rivers/Dams (Protection Of Aquatic Life)

Determinand	**Unit**	**Minimum**	**Median**	**Maximum**
Physical				
Conductivity	mS/m	See notes	See notes	See notes
pH	pH unit	6.0	6,5 - 9,0	9.0
Suspended solids	mg/ℓ	25		80
Temperature	°C	See notes	See notes	See notes
High toxicity				
Cadmium	µg/ℓ	0,1	3	30
Cerium	µg/ℓ		20	
Chlorine (disinfection)	mg/ℓ	0.002	0,003	0,01
Copper	µg/ℓ	5	5	200
Cyanide	µg/ℓ	5	5	5 000
Hydrogen sulphide	mg/ℓ	0.002	0,002	0,3
Lead	µg/ℓ	20	30	100
Lithium	µg/ℓ		5	
Mercury	µg/ℓ	0.05	0.2	10
Pesticide, aldrin	ng/ℓ	1	10	10
Pesticide, chlordane	ng/ℓ	10	25	40
Pesticide, DDT	ng/ℓ	1	1,5	2
Pesticide, dieldrin	ng/ℓ	1	5	5
Pesticide, endosulphane (α+β)	ng/ℓ	3	3	6
Pesticide, endrin	ng/ℓ	2	2	4
Pesticide, heptachlor	ng/ℓ	1	5	10
Pesticide, lindane (γ - BHC)	ng/ℓ	10	15	20
Pesticide, malathion	ng/ℓ	8	100	100
Pesticide, methoxychlor	ng/ℓ	5	20	30
Pesticide, mirex	ng/ℓ	1	1	1
Pesticide, parathion	ng/ℓ	0.4	8	40
Pesticide, toxaphene	ng/ℓ	5		10
Phenols	µg/ℓ	1	1	200
Polychlorobiphenyl	ng/ℓ	1		2
Selenium	µg/ℓ	5		10
Moderate toxicity				
Aluminium	µg/ℓ	100		1 500
Ammonium (as N)	mg/ℓ	0,016	0,016	124
Antimony	µg/ℓ	200		2 000
Arsenic	µg/ℓ	10	200	1 000
Beryllium	µg/ℓ	11		1 100
Bromine (Br_2)	µg/ℓ		100	
Chromium	µg/ℓ	10	50	100
Detergents (as MBAS)	mg/ℓ	0,2		0.5
Fluoride	mg/ℓ	1,5	1,5	1.5
Iron	µg/ℓ	200	200	1 000
Manganese	µg/ℓ	100		1 000
Molybdenum	µg/ℓ		100	
Nickel	µg/ℓ	25	50	50
Pesticide -2,4-D	µg/ℓ	4	4	4
Phthalate esters	µg/ℓ	0,3		3
Thallium	µg/ℓ		100	
Thorium	µg/ℓ		100	
Zinc	µg/ℓ	30	100	100
Low toxicity				
Barium	mg/ℓ	0,5	1	5
Cobalt	mg/ℓ		1	
Iodide	mg/ℓ		1	
Rubidium	mg/ℓ		2	
Tin	mg/ℓ		1	
Uranium	mg/ℓ	0,1		10
Vanadium	mg/ℓ		0.5	

1980 criteria - presently being updated

Table 15 : Water Quality Criteria For Rivers/Dams (Protection Of Aquatic Life) - continued

Determinand	Unit	Minimum	Median	Maximum
Non-toxic				
Alkalinity (as $CaCO_3$)	mg/ℓ	>20	>20	>20
Boron	mg/ℓ	1,5		5
Calcium	mg/ℓ		1 000	
Chloride	mg/ℓ	50		400
Magnesium	mg/ℓ		1 500	
Oxygen, dissolved	mg/ℓ	>4	>5	>5,8
Phosphate, total (as P)	mg/ℓ		0,1	
Potassium	mg/ℓ		50	
Silica (as Si)	mg/ℓ		50	
Sodium	mg/ℓ		500	
Strontium	mg/ℓ		200	
Sulphate	mg/ℓ		1 400	
Titanium	mg/ℓ		12	
Tungsten	mg/ℓ		1	

1980 criteria - presently being updated

NOTES TO TABLE 15:

The criteria are for the protection of fresh water aquatic life. Criteria for electrical conductivity and temperature depend on local conditions and on the life species present. Fish are more sensitive to sudden changes in electrical conductivity or temperature than to the absolute values of these determinands.

The minimum and median criteria for mercury are based on the danger to humans of methyl mercury poisoning on eating mercury contaminated fish. The same applies to phenol, although here it is the phenol taste imparted to the fish fillets which is of concern rather than phenol toxicity.

Dissolved oxygen is particularly important for aquatic life. The oxygen levels should exceed the stated criteria. The toxicity of the metallic elements (lead, zinc, copper, etc.) to fish is higher in low calcium content water or in water with a low conductivity : for such waters, the minimum criteria rather than the median criteria for metallic elements should be used.

5. INTERACTIONS

Various water quality guidelines and standards have been presented in the tables above. The interconnectivity of water quality guidelines and effluent standards (or the receiving water quality objective) is presented schematically in Figure 1.

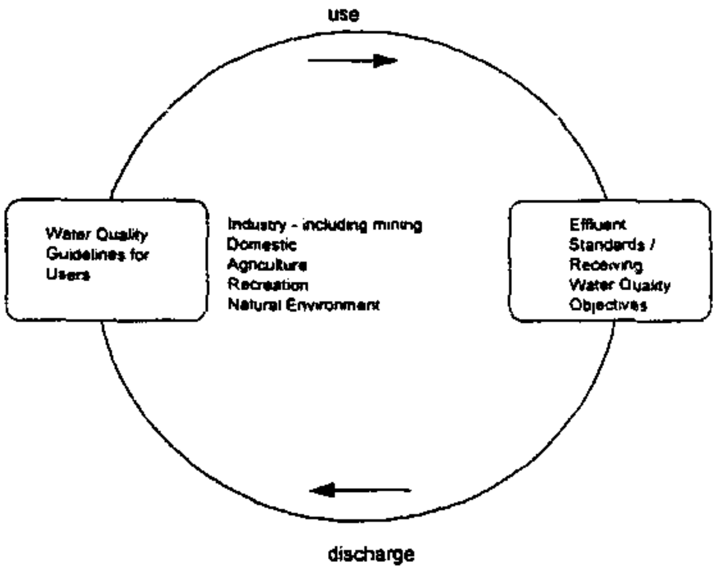

Figure 1 : Schematic Presentation Of The Links Between Water Quality Guidelines And Effluent Standards.

These links indicate that the users of water, which will include mines, have certain water quality requirements, and that during the course of its usage, the water may undergo a change which, should it be discharged, will have to comply with either the general effluent standard or a receiving water quality objective. This water mixes with the main stream and becomes available again for users. The water quality requirements of these users, which may also include mines, will however need to be met. Hence the application of standards to attempt to control the quality of water in the main stream.

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WATER USE AND IMPACTS

1. INTRODUCTION

Water is a fundamental necessity for mining, both in terms of the quantity as well as the quality. The use of water in mining covers a variety of processes, from potable water to cooling water to acting as a transport medium, with many uses in between. Although there is some similarity between the use of water in the gold and coal mining industries, there are also some specific differences.

2. GOLD MINING INDUSTRY

2.1 BACKGROUND

Water in the gold mining industry is used underground in drilling operations, for dust suppression, environmental cooling, condenser circuits on refrigeration plants and in hydropower(7). Additionally, potable water is supplied underground for drinking. In surface operations, water is used to transport the ore after it has been crushed and milled(7). The addition of water allows the ore to be gravity concentrated and thickened; it allows cyanidation to be performed, followed by either filtration or carbon-in-pulp recovery, and it transports waste materials to the slimes dams(7). Potable water is also supplied to the hostels, residential areas and surface plants(7).

The actual volumes of water used varies from mine to mine. Some mines have large amounts of fissure water which need to be disposed of, others have to purchase large volumes of clean water, and others still recirculate much of the water and so have only to purchase make-up water(7).

Pulles (1992), developed a mass water balance for the gold mining industry which is shown schematically in Figure 1.

The gold mining industry consumes and circulates an estimated 73 000 ℓ/s (6 400 Mℓ/day) of water. Approximately- 63 600 ℓ/s (5 500 Mℓ/day) of this water is circulated in closed loops. The bulk of this water (34 700 ℓ/s) is circulated as condenser water for the refrigeration plants, while 9 800 ℓ/s of water is circulated to bulk air coolers to cool the air underground. A further 12 100 ℓ/s of water is circulated for mining purposes, of which 4 800 ℓ/s is chilled water, which performs a supplementary cooling function in the stopes. Taken together, a full 78 per cent of the water in circulation is associated with mine cooling in one way or another. Finally, approximately 7 000 ℓ/s of water is circulated between the reduction plant and the slimes dam for metallurgical purposes(7).

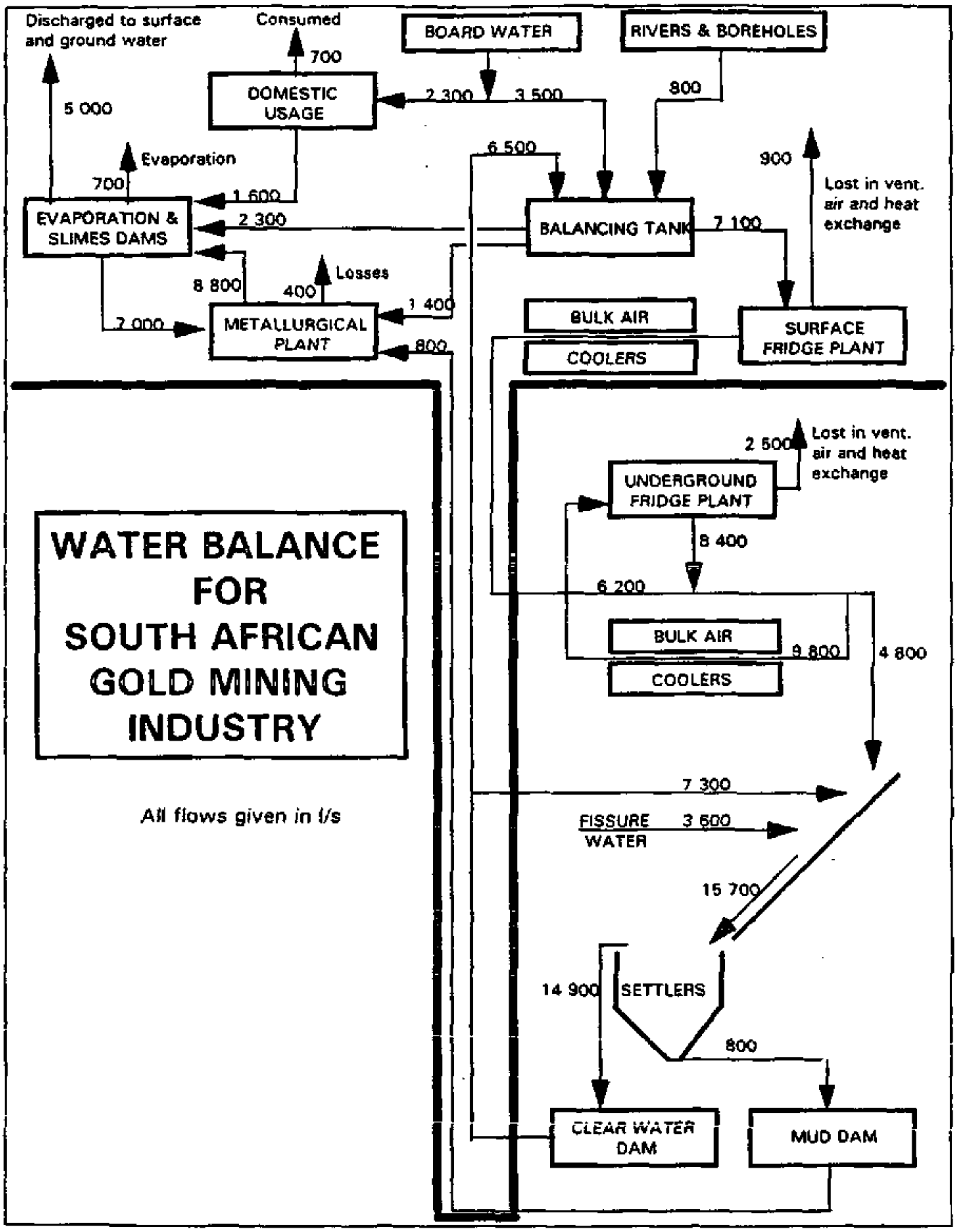

Figure 1: Mass Water Balance for the Gold Mining Industry

In terms of water sources, approximately 70 Mℓ/day of water is abstracted from rivers and boreholes, Water Boards supply 510 Mℓ/day and fissure water accounts for 320 Mℓ/d. Volumes of water discharged account for 440 Mℓ/day and some 460 Mℓ/day is lost through evaporation. Figure 2 presents a summary water balance for the gold mining industry.

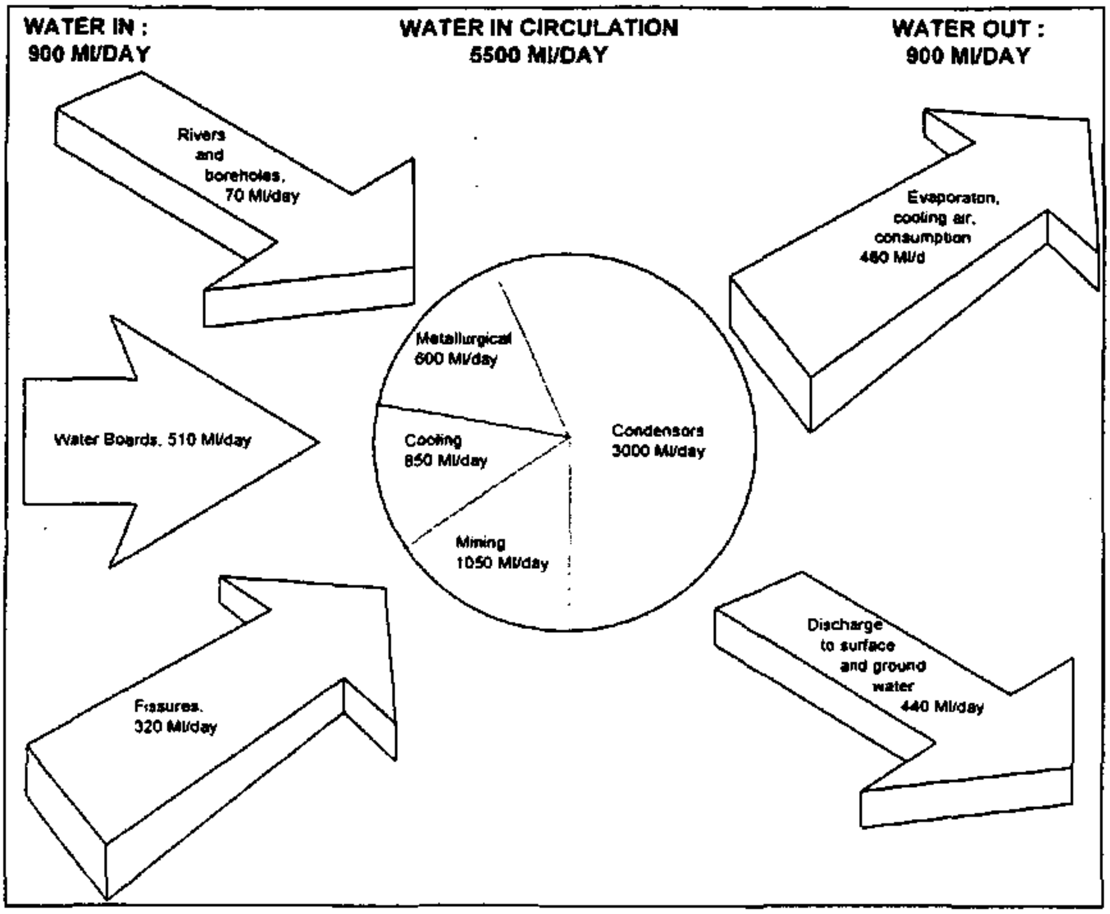

Figure 2 : Summary Water Balance For The Gold Mining Industry.

To place volumes of water used within gold mining in perspective with other users of water, gold mining uses approximately 1 per cent of the total water consumption in South Africa. Table 1 presents water usage for various user groups.

Table 1 : Water Usage For Various User Groups

User Sector	$m^3 \times 10^6$/year	Per cent
Irrigation	10 776	66.2
Stock watering	222	1.4
Nature conservation	219	1,3
Urban & rural domestic	3 829	23.5
Industrial & other mining	586	3.6
Power stations	467	2.9
Gold Mining	181	1.1
TOTAL	**16 280**	**100**

2.2 WATER USE AND WATER QUALITY IN GOLD MINING

2.2.1 General

As described above, water within the gold mining industry is crucial in the extraction of the ore underground and ultimately in the production of gold. It has also been alluded to that water quality is as important in some of the processes. More information concerning this aspect is presented here.

2.2.2 Underground Circuits

The major uses of water underground are for cooling, drilling, water jetting, and the provision of hydraulic emulsions(11). Cost and availability of fresh water have made it necessary to reuse mine service water wherever possible(12). As detailed earlier, some 5500 Mℓ/day are in circulation within the gold mining industry. This circulation of water can result in water quality changes, as a result of the actual mining operations as well as the mixing with poorer quality fissure water which enters the mine. During the mining operation, explosives are used to break the rock. This requires drilling into the rock, to lay the charges of explosives. These drills require cooling. Additionally, a large amount of dust is generated, and this has to be suppressed with large amounts of water(2). The chemicals contained within the explosives also find their way into the spent water circuits(2, 9). The fissure water typically contains relatively high concentrations of salts, which have been leached from the rock. When this mixes with the spent service water, the net result is a deterioration in water quality underground.

This deterioration in quality affects different processes, depending on the specific elements within the water. Table 2 presents information on water use, the main water quality parameters which affect the use and the associated problems.

Table 2 : Water Use, Water Quality Parameters Affecting Use, And Associated Problems.

WATER USE	MAIN WATER QUALITY PARAMETERS	ASSOCIATED WATER USE PROBLEMS
General mine service water	Suspended solids	Increased consumption of chlorine for disinfection; erosion & clogging of orifices and fouling
	Ammonia (from explosives)	Inhibits adequate disinfection
	Acidity	Corrosion; affects disinfection of water
	Microbiology	If disinfection is impaired - potential for cholera, typhoid and dysentery increased.

Table 2 : Water Use, Water Quality Parameters Affecting Use, And Associated Problems... Continued

WATER USE	MAIN WATER QUALITY PARAMETERS	ASSOCIATED WATER USE PROBLEMS
	Radioactivity	Risk at present is estimated to be low - further research required.
	Diesel, fuels, oils from mechanised mining	Accidental spillage and dumping affects settlers; blinds carbon in CIP process; affects chlorine disinfection
Cooling / Refrigeration Systems	Total dissolved salts	Scale formation on heat exchange surfaces; corrosion of pipes;
	Suspended solids	Fouling of heat exchanger tubes
Pumps	Total dissolved salts	Scale formation, corrosion, pitting;
	Suspended solids	Blockage of screens; excessive wear; reduced efficiency;
	Acidity	High acidity leads to increased corrosion; high alkalinity leads to increased scaling
Dust suppression	Suspended solids	Health problems if vapour is inhaled;
	Microbiology	Vapour may contain *Legionella* spp.
Settlers	Suspended solids	Highly variable loading can lead to silting up of clear water dams as a result of impaired settling
	Acidity	High acidity (low pH) interferes with flocculation
Backfill	Suspended solids	Places additional load on settlers
	Cyanide	Potential health risk; may further leach gold which is "lost";

2.2.3 Surface Circuits

Surface water circuits have very different water quality requirements based on the different recovery processes which are used(7). In general terms there are three circuits:

1. process plant water circuits
2. domestic water circuits
3. waste water or effluent circuits.

2.2.3.1. *Process plant water circuits*

Table 3 presents information of process plant water use and water quality requirements[7].

Table 3 : Process Plant Water Use And Water Quality Requirements

Process	Process description	Water quality requirements
Cyanidation	High solution pH required	Water quality not critical
Filtration to recover gold	Separates slime from solution; subsequent precipitation with zinc dust and lead nitrate; precipitate is calcined and smelted	Water quality not critical
Carbon-in-pulp	Gold is absorbed from cyanide leach directly onto activated carbon; carbon separated from slimes; gold eluted from carbon and recovered by electroplating or zinc precipitation	Large organic molecules (e.g. in treated sewage effluent) reduces capacity of activated carbon; high magnesium and calcium in water leads to a fine floc being precipitated; high copper overwhelms normal displacement of copper by gold; Prussian Blue(ferro-ferricyanide) from partial decomposition of pyrite stains activated carbon.
Pyrite flotation	Before extraction - reverse leach; after extraction - forward leach.	Cyanide and lime used to ensure alkaline conditions have adverse effect on pyrite recovery; soft acidic water needed to repulp dewatered slimes; sensitive to organic solvents
Sulphuric acid production	Water is used to cool the roast gases in the Peabody scrubbers	Water must have a low chloride content to prevent excessive corrosion of the stainless steel equipment.

2.2.3.2. *Domestic water circuits*

Large volumes of water are supplied to mine offices and hostels. The hostels utilise between 0,2 and 0,7 m^3/person/day, with approximately 75 per cent of this going to the ablution blocks and the remainder going to the kitchens and beer halls. The water supplied is of potable water quality[7].

Sewage treatment is normally done on site using a conventional sewage treatment works. Best (1985), reported however, that approximately 60 per cent of these works produce poor quality effluents. The reasons for this are:

- overloading, especially during shift changes;
- lack of training and skill of the operators;
- lack of management support and interest.

The water can be reused on the mine in the following areas:

- as a supplement to service water if properly disinfected;
- as irrigation water;
- as replacement water to be discharged to rivers;

2.2.3.3. *Waste water or effluent circuits.*

The fate of waste streams from mines falls into two main categories:

- deposition on the mine in the residue deposits
- active and passive discharge of effluent to the broader environment.

In this section only the deposition on the mine will be considered. The active or passive discharge of effluents will be covered in section 2.2.3.3.2 Water Discharged from Gold Mines.

2.2.3.3.1 Residue deposits

Table 4 presents general information on residue deposit type and the associated water issues.

Table 4 : Residue Deposit Type And Associated Water Issues

Residue deposit type	Water related issues
Rock dumps	Large particle size (small surface area) results in low potential for pyrite oxidation - low pollution potential.
Sand dumps	Small particle size (large surface area) and high permeability results in significant pyrite oxidation; effluent can be acidic, saline, have high concentrations of sulphate, iron and manganese, as well as arsenic, aluminium, cadmium, chromium, cobalt, lead, mercury and nickel.
Slimes dams	Problems similar to sand dumps but two or three times smaller in magnitude.

2.2.3.3.2 Water Discharged from Gold Mines

Water discharged from gold mines falls into two main groups, active and passive. In terms of active discharge, this can be sourced from the sewage treatment works, from dewatering of the mine, and overflows from holding reservoirs and dams. Passive discharge (i.e. not conscious discharge) includes seepage from residue deposits (both to surface streams and to groundwater systems).

These effluent streams require permits from the Department of Water Affairs & Forestry. Their quality is routinely monitored and some of the considerations include pH, total dissolved salts, sodium, ammonia, manganese, cyanide, heavy metals and suspended solids. For sewage effluent, phosphates, nitrates and microbiological components are also included.

3. COAL MINING

3.1 BACKGROUND

Coal is South Africa's main source of energy and is a feedstock for the petrochemical industry[5]. The demand for coal in South Africa is increasing rapidly each year. Certain political influences can be expected, at times, to increase this demand still further.

Water in the coal mining industry is used within the process of extraction (for drilling and continuous mining machines) in dust suppression and for potable use. Coal mining in South Africa is not done to the same depths as gold mining and therefore the need for significant cooling of the mine is reduced. Coal mining is also carried out in open cast operations using blasting, drag lines and truck and shovel operations. Within the coal plants, water is used for washing the coal, for transporting the slurry to the discard dump and for potable consumption.

To date, no detailed mass water balance for the coal mining industry has been determined. The use of water underground is limited, but significant quantities are required for the beneficiation of coal. Water entering the mine underground is usually pumped to surface where is can be used for coal washing. Purified sewage effluent is also used for coal washing(5). It has been estimated that 0.3 m^3 of water is required per ton of coal. This can be compared to approximately 2,5 m^3/ton for gold[5]. From a water quality viewpoint, coal ore contains pyrites which results in a deterioration in water quality when the pyrite is oxidised and acid conditions develop. These acid conditions further influence water quality by leaching salts and metals from the ore[10].

3.2 WATER USE AND WATER QUALITY IN COAL MINING

3.2.1 General

Although the volume of water required for the coal mining industry are less than that required for the gold mines, water is still required for the extraction of coal. The quality of water available is also important, specifically in terms of its potential effect on the equipment used.

3.2.2 Underground Circuits

Water used in underground circuits must conform to the same basic principles as water for the gold mining industry. These are that water used in equipment cooling should not be excessively scaling or corroding, that water which may be inhaled or drunk underground be free of microbial contamination and conform to potable standards, and that any spillage of fuel, diesel or oil should not contaminate the water. Most use of water is for dust suppression(6).

3.2.3 Open cast circuits

Water use in open cast mining is restricted to dust suppression on the haul roads and potable supply. The quality requirements of water to be used on roads is only important if there is a potential to contaminate river systems if the quality is very poor.

3.2.4 Surface circuits

Normally raw coal is fed into crushers (sometimes primary and secondary) and then to a stockpile. Coal is then drawn from here to the beneficiation plant where dense medium separation is carried out. Washing densities are controlled with magnetite. Once washed, the coal is either stockpiled or loaded into coal trucks. Rejects are processed at the thickener plant. This may involve the use of settlers or hydrocyclones to dewater the reject. The reject is pumped to the discard dump for disposal(3).

In the past slurry was typically hydraulically placed in ponds at the base of the discard dump with no real active drainage systems. A newer trend in discard disposal allows for slurry to be deposited all around the discard dump. The water is then reclaimed using penstocks and under drains.

3.2.5 Water discharged / seepage from Coal Mines

There is normally a high degree of water reuse within the coal mining operations. However, water can leave the mine in the following ways:

- in the coal for distribution
- overflow from pollution control facilities
- seepage from discard dumps, stockpiles, rehabilitated open cast pits and defunct coal mines
- discharge of treated sewage effluent if not used within the mining process.

There are a number of water quality constituents which may be in this water which influence the receiving river systems:

- coal fines in the water which on settling can silt up dams and weirs, and which also provide a source of salts and other elements which may deteriorate water quality

- seepage from discard dumps and defunct mines are often acidic and normally have high concentrations of salts and metals as a result of pyrite oxidation(1). Elements of concern are sulphates, sodium, chloride, aluminium, fluoride, cadmium, nickel and other heavy metals(4).

4. IMPACTS OF MINING ON WATER QUALITY

4.1 TYPICAL IMPACTS

The nature of gold and coal mining leads to a change in water quality. How this change affects the use of this water downstream, depends on what has or has not been done to the water. As outlined above, different processes and waste streams have different water quality profiles.

Those elements of concern to the natural environment and which are sourced from mining activities are summarised in Table 5 along with a summary of the affected natural systems.

4.1.1 Total Suspended Solids (TSS)

Total suspended solids are important in freshwater systems because they influence light penetration into water. Light is necessary for photosynthesis, which is the basis of all nutrition in the food chain. Total suspended solids (TSS), more than TDS, interfere with penetration of light by scattering and absorbing its rays. Turbidity is an optical property of water and is not accurately indicative of the weight of suspended material in a given amount of water, because suspended particles differ in their properties. Apart from scattering and absorbing light, (and the subsequent effects on photosynthesis) suspended matter may be injurious to aquatic life. Certain particles may be abrasive, although some are more harmful than others.

An EDAX scan of the TSS pumped from underground mine service water showed the following composition :

ELEMENT	Avg.
Sulphur	23,74 %
Chloride	0,05 %
Calcium	36,45 %
Magnesium	6,25 %
Silica	6 %
Iron	25 %
Manganese	1,75 %
Nickel	1,25 %

As can be seen, the proportion of the elements was Ca > Fe > S > Mg > Si > Mn > Ni > Cl. Out of these elements Ca is the most abundant and Ca + Fe + S make up about 80 % of the TSS. An important factor here is that Ca and SO_4

are associated and probably saturated, while Fe would probably occur in the Fe(III) form.

Table 5 : Contaminants, sources and areas affected

Contaminant	Typical source	Areas affected
Metals Iron Manganese Zinc Lead Copper	Pyrite oxidation in underground stopes & surface rock and sand dumps & slimes dams and coal discard dumps, rehabilitated opencast pits and defunct mines with dissolution of metals;	Sediment, groundwater, surface waters, macrophytes, biota
Sulphate	Pyrite oxidation in underground stopes & surface rock and sand dumps & slimes dams and coal discard dumps, rehabilitated open cast pits and defunct mines with production of sulphates	Sediment, groundwater, surface waters
Cyanide	Spillage from: plant areas, ruptured slimes delivery pipeline and slimes dams	Sediment, groundwater, surface waters, macrophytes, biota
Suspended solids	Inadequate underground settling, runoff from surface rock, sand dumps & slimes dams and coal fines	Sediment, groundwater, surface waters, biota
Sodium	Fissure water, addition of sodium based neutralisation chemicals	Sediment, groundwater, surface waters, macrophytes, biota
Chlorides	Fissure water	Sediment, groundwater, surface waters, macrophytes, biota
Nitrogen compounds	Waste explosives in underground stopes, gaseous by-products from explosives, sewage and contaminated run-off from hostels	Groundwater, surface waters
Phosphates	Sewage and contaminated run-off from hostels	Groundwater, surface waters
Acidity	Pyrite oxidation (underground, surface dumps (rock, sand & slimes), spillage from plant areas and coal discard dumps, rehabilitated open cast pits and defunct mines)	Groundwater, surface waters, macrophytes, biota
Radionuclides	Pyrite oxidation in underground stopes & surface rock and sand dumps & slimes dams with dissolution of radionuclides	Sediment, groundwater, surface waters, macrophytes, biota
Microbes Faecal coliforms Coliphages	Faecal contamination of u/g mine service water, poorly treated sewage, run-off from hostel areas, livestock grazing	Sediment, groundwater, surface waters, macrophytes, biota

Settling facilities and pH stabilisation programs may be implemented to reduce any negative impacts. Settled sludge from settling ponds and settlers should be disposed of on active slimes dams.

Because of the reduced buffering capacity of certain mine waters, the pH is prone to sudden decreases that will cause more soluble elements like Fe(II) to be dissolved into the water while Fe(III) would have settled out.

The pH is not only an important indicator of acidity or alkalinity, but it has secondary influences on the metals and therefore the TDS and E.C.

4.1.2 The pH, Carbonate and Hardness System

One of the major problems in the water from mine dumps as well as from water pumped from shafts and stopes, is the acidity as a result of acid mine drainage.

The acid mine drainage is a result of Pyrite (FeS) reacting with water and oxygen in the presence of bacteria. In this reaction

$FeS + H_2O + O_2$ (+ Bacteria) -> $H_2SO_4 + Fe$

sulphuric acid is produced and the pH can drop to as low as 1,5[13]. The low pH not only causes many metals to go into solution, but the associated high concentrations of sulphates are a major source of salinisation of natural waters. Many impacts (except nutrient loading) are therefore associated with this problem. Research is currently underway to address the problem at the source, in the stopes.

Research into reducing nutrients and restoring the acid buffering capacity of water through constructed wetlands, is now also underway.

In a wetland, gas exchange takes place between the emergent plants and the atmosphere and so a limited amount of carbon dioxide is fed into the water. Carbon dioxide is important in maintaining the carbonate balance of the water. Algae and lower organisms, though, respire in water and therefore increase carbon-dioxide concentrations in the water. The fungi and bacteria that assist in the decomposition of organic matter are another source of carbon-dioxide in water. Once the biota start dying, the natural processes such as photosynthesis and decomposition (which usually sustain the carbonate balance in aquatic environments) are limited, and the buffering capacity of the water decreases. The lower the buffering capacity becomes, the easier the pH of the water is decreased by acid mine drainage. This causes a positive feedback mechanism in the ecosystem[14] (which means a snowball effect in which the condition of the system increasingly deteriorates). Eventually the balance in the aquatic ecosystem is disturbed to a point where it cannot recover by itself.

By introducing mine effluent, containing high nitrate concentrations into a specifically constructed wetland, algae and macrophyte growth is stimulated, resulting in photosynthesis that increases pH buffering capacity (raising the low pH of the acid mine drainage) and converting nitrates, nitrite and ammonia into gaseous nitrogen released into the atmosphere with no influence on the environment.

This will help the biological filtering effect of the natural ecosystem to maintain itself.

4.1.3. Total Dissolved Solids (TDS)

The effect of the high TDS on the aquatic organisms includes disruption of many physiological processes. Most organisms in the fresh water aquatic and semi-aquatic environment are adapted for osmotic conditions so they can excrete osmotically the salts they do not require(15). Water tends to move by means of diffusion from a high concentration of water to a lower concentration. In fresh water, the plants and animals contain more salts in their tissue/cells than the surrounding water. The water concentration in the organisms is thus lower than in the surrounding water medium. Water will osmotically move into living tissue until a balance is reached. Fresh water organisms are adapted to compensate for the continuous influx of water into their cells. When the salts, indicated by TDS, in the water are very high (as in the case of most seepage and water affected by acid mine drainage), the water in the organisms osmotically leaves the organisms. Physiological processes, such as ion uptake and excretion in the plants and animals are disrupted in this way and death may occur.

The salts which contribute the greater percentage of TDS are fortunately not toxic and have no adverse health affects. The salts that are the main driving force behind high TDS and electrical conductivity (EC) values in mine water, will be discussed in the following section.

4.1.4. Salts

As far as sulphate, chloride and fluoride are concerned, fluoride is probably the constituent that needs to be monitored closely, as it can have health implications at high concentrations. It is however a necessary trace element for humans. Chloride concentrations in the Transvaal shales can become a problem for irrigation, but no health risks to the users are expected. Chloride usually associates with the alkali metal sodium.

Sulphates, which are a product of acid mine drainage, may contribute up to 50% of the TDS or salinity loading.

4.1.5. Alkali Metals

Sodium concentrations are significant in the Free State Goldfields areas. This element can have a significant impact on the use of the water for irrigation, as a high sodium absorption ratio can cause serious problems with infiltration of water. The relationship of calcium and magnesium to sodium determines the sodium absorption ratio. Sodium is physiologically important to many biota in the same way as some alkali earth metals such as magnesium and calcium.

4.1.6. Alkali Earth Metals

Calcium and magnesium have a limited influence on most users of water. Calcium and magnesium both occur in dolomite, which is a very common sedimentary rock on the Witwatersrand. Calcium tends to bind with sulphate and precipitate, but very high values of calcium and sulphate indicates supersaturation even though most of the TSS consist of calcium sulphate that will eventually settle out. The potential for calcium and magnesium to form scale may influence certain uses susceptible to scaling.

4.1.7. Metalloids

Silica as a metalloid has a limited affect that is probably only a secondary response to very acidic conditions. Reducing acidity could therefore alleviate this limited affect. Silica is not exceptionally toxic and therefore its impact on the environment is limited.

4.1.8. Metals

Metals do pose a potential impact to down stream users, and it may not always be strongly correlated with very low pH situations. Lead and manganese seem to persist in the drainage systems in spite of alkaline pH values. Other metals like copper bond with cyanide, thereby reducing the potential impact.

Passive oxygenation and temperature reduction as well as active pH control and settling should reduce the potential impact of metals. Once again sludge, from settling facilities, needs to be handled with care and deposited on well constructed slimes dams.

The solubility and toxicity of most metals is pH dependant and it is expected that where acid mine drainage occurs, toxic metals will be mobilised into biotically absorbable forms. Not only are some of the metals toxic, but combinations of lower concentrations of metals together have a synergistic effect on the biota.

Cadmium is a metal that could cause carcinogenic conditions in humans when ingested regularly at specific concentrations. Standards for aquatic life protection are set at 30 µg/ℓ.

Copper is also potentially toxic and concentrations of less than 5 µg/ℓ are suggested for aquatic life protection with an absolute maximum of 30 µg/ℓ. Copper solubility is also pH dependant, and as with most metals, it becomes soluble in water at low pH values. Cyanide-copper bonds can also lead to soluble forms of copper in water, increasing the concentrations.

Iron is a less toxic metal than most of the others, but its abundance and high concentrations can cause it to have a significant influence on the environment.

Manganese, which is soluble in acidic water as manganese(II), is usually decreased by the lime that is added for pH stabilisation, although pH values of 9-10 are required to significantly reduce manganese levels.

Zinc is another metal, that is used in the gold extraction process, and it too, has potentially toxic effects.

An example of the effects of one of these metals is lead, which causes the inhibition of an enzyme that is responsible for the formation of haemoglobin in blood. Furthermore, lead from dietary sources accumulates in the skeleton of man. Calcium ions are replaced by lead and the half life of these lead ions is estimated at 2-3 years. High concentrations in the blood and body become fatal. Lead nitrate is often used in the gold extraction process.

4.1.9 Nutrients

This is a group of compounds that are organic or food substances for some plants and micro organisms. Nitrates from explosives in the ore residue fluctuate depending on the type of explosive used, from 25 - 75 g/ton.

Nitrates may be a problem from mines, but they can be effectively reduced by wetland systems, thereby reducing eutrophication in receiving water bodies. Nitrates can further pose a serious health risk for human infants(16).

Other sources of nitrates - and nutrients in general - are sewage facilities on the mines. With appropriately designed and operated sewage plants, nitrates can be removed effectively at low cost, and phosphates can be removed biologically or chemically.

Phosphates have a significant effect on eutrophication. Eutrophication can lead to low oxygen concentrations at which aquatic biota, including fish, could die. On mines, the major source of phosphates is treated sewage effluent. On mines where activated sludge is used as secondary sewage treatment, phosphates may be reduced significantly. In addition, wetlands may be used to remove phosphates from effluent that will be discharged.

In a similar manner, nitrate concentrations can indirectly increase pH values through eutrophication, and this usually decreases metal mobilisation (except lead in this case). This however decreases oxygen concentrations which reduces metal precipitation.

Ammonia (NH_3) usually occurs in acidic water, as opposed to ammonium (NH_4) in alkaline water. The relationship is also temperature dependant. Ammonia toxicity is pH dependant and can become toxic to biota, even though it is a nutrient to specific bacteria (like *Nitrosomonas* and *Nitrobacter*).

4.1.10 Cyanide

The toxicity and removal / treatment of cyanide is discussed in a separate literature review.

4.1.11 Radionuclides

The toxicity and removal / treatment of radionuclides is discussed in a separate literature review.

The determination of an impact requires that the quality and volume of the water is known, as well as the identified downstream users of the water.

4.2 RECOGNISED USERS

The Department of Water Affairs & Forestry has identified 5 legitimate water user groups:

1. Domestic
2. Industry
3. Agriculture
4. Recreation
5. Natural Environment

Specific water quality guidelines for each of these user groups have been developed (except Natural Environment) and are published as a series by the Department of Water Affairs & Forestry. They provide a useful starting point on which impacts can be assessed.

4.3 SCOPE OF IMPACTS

The scope of impacts from mining operations are as important as the nature of the impact. Impacts are assessed against local and regional conditions. This results in certain impacts being significant in the immediate vicinity but insignificant in the wider area. The importance of this can be seen in the impact of cyanide on the environment. Locally, around active slimes dams, there is a limited impact, but some distance away from the slimes dam, there is no impact.

Additionally, if a specific user is identified at a point some distance downstream of the mine, the influence of the mine on that user group should not be considered within the mine property. The net effect, is to restrict the assessment to only those users that are recognised as being impacted.

5. SUMMARY

In this review, an assessment has been presented on the use of water in the gold and coal mining industries. Additional information has been presented on the use of water in specific processes and the water quality requirements of these processes. The typical water quality problems associated with discharges (active and passive) of water from these industries and the scope of related impacts have also been presented. The site specific nature of the impact of mining and additional detail regarding impact assessments can be found in the WRC document " A manual to assess and manage the impact of gold mining operations on the surface water environment"[8].

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ACID MINE DRAINAGE

1. INTRODUCTION

The generation of acid from freshly exposed sulphidic rock can occur wherever sulphide bearing ores are brought into contact with air and water, usually as a result of some disturbance to the geological environment. The most common geological disturbances are due to mining and the term acid mine drainage has been coined to describe this phenomenon. Potential sources of acid mine drainage include surface runoff from open cast mining areas, runoff from residue dumps or ore stockpiles and drainage from underground workings. In South Africa the problem is found on both coal and gold mines with the extent of the problem varying from mine to mine although, generally, the most severe problem is experienced in active mining areas where fresh material is constantly being exposed. The problem manifests itself with the formation of acidic water which then requires the application of a pH neutralisation program in order to prevent the circulation of a very corrosive type of water. The pH control program implies a chemical cost (usually lime) as well as a scaling or corrosion problem (cost) in the circulating pipework.

2. ACID GENERATION

Not all sulphide-bearing material will necessarily produce acid mine drainage when exposed to air and water and the correct micro-environment must be present for the acid generation process to occur.

It is generally well established and accepted that the major factor contributing to the formation of acid mine drainage is the presence of the bacteria *Thiobacillus ferrooxidans* and *Thiobacillus thiooxidans*. Although the collection and subsequent treatment of acid mine drainage is the subject of another review, one of the strategies employed to minimise the impact of such water on receiving environments is to prevent or control the development of the acid, either at the generation stage, or the migration stage. With this in mind, and considering the importance of the bacterial role in the process, a brief discussion of the bacteria physiology and mechanisms of oxidation are given. An understanding of these factors is essential in understanding the possibilities for control.

In addition, although the bacteria are able to oxidise the following sulphide minerals: arsenopyrite, bornite, chalcopyrite, covellite, enargite, galena, marcasite, millerite, molybdenite, orpiment, pyrite, stibnite, sphalerite and tetrahedrite[1] the major acid mine drainage problems in South Africa are caused by pyrite oxidation, and this discussion will thus be limited to dealing with the mechanisms of bacterial attack on pyrite. To this end, the mechanisms of attack on the sulphur component and the ferrous component will be dealt with separately.

2.1 PHYSIOLOGY OF BACTERIA

All members of the genus *Thiobacillus* use reduced sulphur compounds such as thiosulphate as electron donors and carbon dioxide as a carbon source. They are unable to use organic compounds and are therefore strict autotrophs.

Thiobacillus thiooxidans was isolated by Waksman and Joffe in 1921 from soil. This bacterium grows best below pH 5 and has been known to produce a negative pH. Aside from it's acid habitat, this species is mainly distinguished by it's ability to oxidise elemental sulphur at a rate comparable to it's oxidation of thiosulphate.

Thiobacillus ferrooxidans was discovered and named by Colmer and co-workers in 1950. *T. ferrooxidans* is morphologically identical to *T. thiooxidans*. In addition to sulphur compounds, this species is mainly distinguished by it's ability to utilise ferrous iron as it's energy source and operates in the same pH range as *T. thiooxidans*.

The ecology of *T. ferrooxidans* and *T. thiooxidans* is similar and associations of these two types of bacteria are common. *T. ferrooxidans* oxidises metal sulphides much more readily than *T. thiooxidans* and therefore tends to be dominant in acidic mine drainage(2). Sulphur oxidation by *T. thiooxidans* results in increased acidity and thus contributes to the occurrence of *T. ferrooxidans* by generating favourable environmental conditions.

The growth requirements of these two species are as follows: an excess supply of oxygen and carbon dioxide, a pH range of 0,5 to 6,0 (optimum 2,0 to 3,5), a supply of ammonium and phosphate ions, a substrate of reduced form of sulphur (thiosulphate, trithionate, tetrathionate, elemental sulphur or sulphide), ferrous iron for *T. ferrooxidans* and an aqueous or at least very humid environment. Optimum temperatures are between 29 and 32°C (3). Although the bacteria function over a wide range of pH, they actively maintain intercellular pH at 6,5 (4, 5).

2.2 REACTIONS INVOLVED IN ACID GENERATION

The overall reaction describing pyrite oxidation is usually written as:

$$4FeS_2 + 15O_2 + 14H_2O \longrightarrow 4Fe(OH)_3 + 8H_2SO_4 \qquad (1)$$

The actual pathway is a series of reactions passing through a number of intermediates. Under natural conditions, the two main oxidising agents acting on pyrite are oxygen and the ferric ion.

$$2FeS_2 + O_2 + 4H^+ + \longrightarrow 2Fe^{2+} + 4S^0 + 2H_2O \tag{2}$$

$$4Fe^{2+} + O_2 + 4H^+ + \longrightarrow 4Fe^{3+} + 2H_2O \tag{3}$$

$$Fe^{3+} + 3H_2O \longrightarrow \underline{Fe\ (OH)_3} + 3H^+ \tag{4}$$

$$FeS_2 + 2Fe^{3+} + \longrightarrow 3Fe^{2+} + 2S^0 \tag{5}$$

$$2S^0 + 12Fe^{3+} + 8H_2O \longrightarrow 12Fe^{2+} + 2SO_4{}^{2-} + 16H^+ \tag{6}$$

$$2S^0 + 3O_2 + 2H_2O \longrightarrow 2SO_4{}^{2-} + 4H^+ \tag{7}$$

Reaction (3) is very slow and in the acidic pH region has a half time of 1000 days (6). Micro-organisms such as *T. ferrooxidans* can increase the rate of oxidation by a factor of one million (6, 7, 8). Reaction (7) may also be bacterially catalysed. The reactions generally found in literature to describe bacterial oxidation of pyrite are:

$$2FeS_2 + 7O_2 + 2H_2O \longrightarrow 2FeSO_4 + 2H_2SO_4 \tag{8}$$

$$4FeSO_4 + 2H_2SO_4 + O_2 \xrightarrow{\text{bacteria}} 2Fe_2\ (SO_4)_3 + 2H_2O \tag{9}$$

$$7Fe_2(SO_4)_3 + FeS_2 + 8H_2O \longrightarrow 15FeSO_4 + 8H_2SO_4 \tag{10}$$

2.2.1 Oxidation of Ferrous Iron

T. ferrooxidans is able to derive energy from the oxidation of ferrous to ferric iron. This oxidation process can be seen from two viewpoints - the chemical aspect and the microbiological aspect. The chemical aspect covers the effects of oxidation in terms of the resultant increased acidity in the environment, increased mineral leaching, etc. The microbiological aspect deals with factors such as the mechanism of oxidation at bacterial level, nutritional and physical requirements, etc. and although not directly measurable, is equally important in obtaining an understanding of the bacteria and considering possibilities of control.

2.2.1.1 *Chemical aspects of ferrous iron oxidation*

Ferrous iron is released initially in reaction (2) either by simple dissociation of the pyrite or by oxidation of the pyrite by oxygen (an electrochemical corrosion reaction). After the sequence has been initiated, a cycle is established in which the ferrous ions released from pyrite are oxidised bacterially to ferric ions, which can then oxidise pyrite, generating more ferrous ions and leading to massive production of acid as seen in reactions (5+6) and (10).

Reaction (2) is very slow. At a pH value above 3,5 the ferric ions produced are removed from solution by precipitation of ferric hydroxide and so reactions (5) and (6) are not important under these conditions. The oxidation pathway then follows reactions (2), (3), (4) and (7). At a pH of less than 3,5, reaction (3) is bacterially catalysed and the cyclical system develops. If the pH remains high enough for iron to precipitate, then 4 moles of acid are released per mole of pyrite, while at the lower pH, the rate of acid production is much higher.

The bacterial catalysis of ferric iron has been thermodynamically formulated (9) and the equilibrium of the reaction can be expressed as

$$\frac{[Fe^{3+}]}{[Fe^{2+}] \cdot [H^+] \cdot [O_2]^{0.25}} = 10^{7.7} \qquad (14)$$

This equilibrium is valid at pH below 3 as precipitation of ferric iron becomes appreciable at pH above 3.

During the bacterial oxidation of pyrite, high iron oxidation ratios (ferrous iron/total iron) above 90% are maintained, coincident with the enhanced release of iron from pyrite. In the absence of *T. ferrooxidans*, a major portion of the iron is in the ferrous form and iron release is not promoted(10). It is generally agreed that *T. ferrooxidans* plays an important role in the oxidation of pyrite through the regeneration reaction of ferric iron. With a high pyrite content, the rate limiting step is the oxidation of ferrous to ferric iron, whereas with a low pyrite content, the rate limiting step is the oxidation of the pyrite (valid in field situation with pyrite of 3,5%). (8)

2.2.1.2 *Microbiological aspects of ferrous iron oxidation*

The mode of action of *T. ferrooxidans* on iron-containing sulphide minerals is not clear, although several possibilities exist. These possibilities include oxidation of the minerals by acidic ferric sulphate with bacteria oxidising the resultant ferrous iron; direct action on the ferrous iron of the sulphide mineral; direct action on the sulphide portion of the mineral or a combined action involving two or more of these alternatives. Both the direct and indirect mechanisms have their proponents, with both camps claiming experimental verification of their proposed mechanism, while some state that both mechanisms occur concurrently (11).

Significant enhancement of leaching rate (10 - 26%) by *T. ferrooxidans* upon adding surface active agents at concentrations between 0,005 - 0,001% (w/v) has been reported (2, 12).

It is claimed that when *T. ferrooxidans* cells are absorbed on the pyrite surface, the iron-oxidising activity is strongly inhibited, but that when the absorbed cells are released into an aqueous phase, the iron-oxidising activity is recovered and pyrite oxidation is markedly promoted (13). Apparently the bacterial concentration in an aqueous phase rather than on pyrite particles, plays a major role in the enhanced oxidation of pyrite (13, 14).

It is possible that the attachment of bacteria on the pyrite surface plays a role in the oxidation of the reduced sulphur compounds, but that ferrous iron oxidation functions best in an aqueous phase as a dissolved substrate is used. The sulphide minerals themselves are not soluble.

Bacteria require a moist environment as a vehicle for transferring nutrients, oxygen and carbon dioxide and removing metabolic products. In order to complete the oxidation reactions, oxygen is required as the terminal electron acceptor. On this basis, a thin film of moisture which permits rapid dissolved oxygen transfer is far preferable to a submerged situation. The same holds for carbon dioxide which is dissolved into the water from the atmosphere. This would suggest that optimal leaching occurs on pyrite particles lying around on the ground in a humid environment (such as in the stopes of a gold mine), rather than on particles submerged in the water.

2.2.2 Oxidation of Sulphur Compounds

All the thiobacilli can derive energy for growth from oxidation of reduced inorganic sulphur compounds. A variety of sulphur compounds can be utilised, although the usual substrates are either elemental sulphur or thiosulphate, with thiosulphate being preferred. The sulphur compounds of interest regarding pyrite and other oxidisable minerals are the metal sulphides. There is as yet no clear picture of the mechanism of oxidation and there is much controversy over even the essential features of the oxidative pathway.

The suggested pathways that are proposed for oxidation of the various sulphur compounds are described in detail in the literature.

- Oxidation of sulphides(15, 16)
- Oxidation of elemental sulphur(16, 17, 18, 19)
- Oxidation of thiosulphate(2, 3, 16, 17, 18, 20, 21, 22, 23, 24, 25, 26, 27)

2.3 FACTORS AFFECTING RATE OF ACID GENERATION

There are various physical, chemical and biological factors which affect the role of bacterial leaching of sulphide ores and the rate of acid generation. These are:

- **Particle size of the ore**. It is generally held that the rate of leaching increases for an increase in specific surface area or a decrease in particle size(28, 29).

- **Oxygen and carbon dioxide supply.** The requirement for a sufficient supply of oxygen and carbon dioxide is obvious since the bacteria are aerobic and their only carbon source is carbon dioxide. This requirement also implies that thiobacilli will proliferate on pyrite particles covered by only a thin moisture layer as found in gold and coal mines.

- **Concentrations of bacterial inhibitors.** Compounds such as sodium fluoride and sodium cyanide inhibit ferrous iron oxidation.
- **Presence of certain metals.** Metals such as Ag, Mo, U and Cu negatively affect the growth of *T. ferrooxidans*(30, 31).
- **Availability of inorganic and organic nutrients, etc.** Various nutrients such as ammonia as a nitrogen source and phosphate for energy metabolism purposes are required by the bacteria. Other required nutrients are potassium, calcium and magnesium.
- **Oxidation-reduction potential.** This determines the activity of the sulphide ore.
- **pH.** pH affects the solubility of metals as well as being a critical factor in the bacterial survival.
- **Temperature.** Temperature, as with pH, affects both the metal solubility and the bacterial activity.
- **Adsorption and ion-exchange capacity of the ore.** These factors influence the mechanics of the oxidation process.
- **Supply of ferrous and ferric iron.** The rate at which ferrous iron is oxidised is related to the availability thereof.

It is clear that most of these rate affecting factors are directly related to the degree of bacterial activity. They may only be present at the source of acid generation or, in the event of the acid water migration, may come into play en route to the environment.

3. DEVELOPMENT AND MIGRATION OF ACID

Once the acid has formed it will either remain and accumulate on the rock or, in the presence of water movement, it will be flushed away. It is usually only when migration occurs that the impacts of the acid drainage become apparent.

During the acid water movement it may encounter an acid consuming or neutralising mineral. The most common of these is calcite ($CaCO_3$) and neutralisation occurs via the reactions:

$CaCO_3 + H^+ \rightarrow Ca^{2+} + HCO_3^-$ or

$CaCO_3 + 2H^+ \rightarrow Ca^{2+} + H_2CO_3$

If acid generation continues long enough at a sufficient rate to exhaust any neutralising ability the pH will continue to drop and create more favourable conditions for acid generation, thus accelerating the rate. The drop in pH may occur in a number of discrete steps as various neutralising minerals become soluble at different pH ranges. This discrete step degradation of pH has been studied and modelled by Morin(32) as well as Morin and Cherry (33). This aspect of mineral dissolution highlights the importance of correct kinetic testing methodology when attempting to predict the long term extent of acid drainage.

As the pH of the water drops, its ability to mobilise certain metals is increased and it is often the elevated concentrations of these metals that has the most significant effect on the environment. These metal loads are usually dropped out of solution when neutralisation occurs, either naturally or due to active treatment.

4. PREDICTION AND MODELLING OF ACID MINE DRAINAGE

In attempting to predict the potential of material to produce an acid drainage, it is necessary to identify the potential for acid generation as well as the potential for neutralisation. This is achieved by the implementation of static testing on representative samples. Once the material has been identified as potentially acid generating the reaction rates of the generation and neutralisation reactions may then be compared by results from kinetic testing. This testing may be carried out using various methods under controlled conditions at laboratory scale or on a larger scale on site.

The information obtained from the test work provides input to a number of mathematical models which can predict the extent of acid generation but the applications are limited to waste rock(34),(35),(36),(37),(38),(39) and tailings dumps(40). Ziemkiewicz(41) describes a spreadsheet model which uses the generally accepted method of acid base accounting to predict loading, concentration and duration of potential acid problems.

Acid generation in the underground mine environment has not been investigated to the same extent as surface dumps. The conditions are vastly different for the two environments and different factors become rate limiting.

5. CONTROL OF ACID MINE DRAINAGE

It is preferable to control acid mine drainage by preventing the occurrence thereof at the source. Where this is not feasible, it may be possible to prevent or control the migration of the contaminated water and thereby reduce the impact on the receiving environment. As a final measure it may be necessary to collect and treat the acid mine drainage once it has reached the receiving environment. Usually this is the only practical option available and as such it is the most widely used approach. This option includes neutralisation in passive treatment systems and such systems are discussed more fully in the relevant literature review.

Whereas pyrite oxidation in dumps is limited by the availability of oxygen, nitrogen and carbon dioxide for the bacteria and water to transport the oxidation products, no limiting conditions are present in stopes. In fact, if the requirements for optimal bacterial oxidation of pyrite are considered, it becomes evident that conditions in underground stopes are close to optimal for pyrite oxidation.

However, a knowledge of the factors which drive pyrite oxidation also indicates that there are a number of variables which, if controlled, could lead to a significant reduction in pyrite oxidation. Many of these potential control variables can be practically controlled in underground stopes without hindering mining operations - in fact, many of the potential control strategies are variations on the conventional drill and blast mining method. Certain new mining methods show promise of reducing pyrite oxidation and include the following:

- hydrocleaning
- continuous scrapers
- non-explosive mining
- hydropower
- rapid face advance mining

There has been significant past research in the South African gold mining industry aimed at inhibition of pyrite oxidation and this includes:

i. The effects of sodium lauryl sulphate (SLS) as a bactericide for use underground were investigated by COMRO in two separate studies(42). Heaps of broken rock were dosed with SLS underground at Durban Roodepoort Deep Gold Mine and it was found that the addition of SLS led to a decrease in bacterial numbers, reduced pH drops across the heaps and reduced sulphate generation on the first wash. It was also found that different explosives led to different sulphate generations.

ii. In a study undertaken by COMRO at West Rand Consolidated Gold Mine, a whole shaft water system was dosed with SLS. Although this shaft system was characterised by large inflows of acidic drainage from old workings, the addition of SLS led to a reduction in the quantity of sulphates being generated in the shaft. This trend was, however, not evident in the working stopes which were more closely monitored. The reasons for this paradox are not known, although adsorption of SLS to the ore in the stopes may have contributed to a reduced effectiveness. It was concluded from this study that research is required on a more fundamental level in order to obtain a better understanding of the mechanisms of this bacterial oxidation before attempts can be made to combat this problem.

iii. In a study undertaken by Gold Fields at Deelkraal Gold Mine, the whole mine water system was dosed with a bacteria inhibiting compound, leading to a significant reduction in soda ash required underground for neutralisation. Although this study evaluated the change in soda ash consumption by the mine, no detailed studies were made to determine the precise effects on pyrite oxidation in the stopes.

In the coal mining industry the use of SLS, sodium benzoate and sorbic acid was investigated as a means of acidification inhibition in coal waste dumps at Witbank(43) Although SLS has previously been used with success for such an application, particularly in the United States(44), the study revealed that *T. ferrooxidans* apparently developed an increased resistance to SLS and recorded the development of microbial populations capable of degrading the detergent. In addition, SLS showed a strong adsorption to coal waste which limited the effectiveness. Smaller scale experimentation indicated that sodium benzoate and sorbic acid may be better inhibitors for coal waste. Sodium benzoate has the added advantage of being much cheaper than SLS.

Control measures for surface dumps are mainly aimed at eliminating the infiltration of water and air. This is achieved by applying various forms of covers ranging from cementatious(45), soil(46), and engineered linings. The disposal of reactive tailings into a sub aqueous environment such as a lake, in order to exclude oxygen, is practised at various mines in Canada but the possibility of a similar disposal method for waste rock has not yet been investigated(47).

6. CONCLUSION

The topic of acid mine drainage and, in particular the behaviour and activity of the relevant bacteria, has been extensively studied. This, however, does not imply that the phenomenon is particularly well understood and, as discussed, there are many varied theories on the exact mechanisms of bacterial oxidation. In addition, the variation in environmental conditions both locally and over time make the prediction of acid mine drainage impacts very difficult. For similar reasons, the prediction of the effectiveness of control measures is equally difficult and any implementation of such measures is site specific. Past research has focused on drainage emanating from surface sources such as waste and residue dumps. The mechanisms and control of underground acid drainage have not been studied in detail and attempts at reducing the impact from these sources are largely a result of extrapolation from surface dump and laboratory investigations. The treatment of acid mine drainage is discussed in other literature reviews.

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CHEMICAL CONTAMINANTS RELATED TO MINING

1. INTRODUCTION

The mining industry has a major role to play in the utilisation and preservation of South Africa's limited water resource.

During active mining operations water is used in underground operations for dust suppression, ventilation/cooling of work areas and refrigeration of service water.

At some mines high pressure water or water jetting is used to remove fine material with a high gold value content from underground workings. Hydro-power is also becoming a power source for drilling and supporting of stopes. In metallurgical plants on surface, water via an extraction process, is primarily used as a transport medium for the milled ore and deposited on the tailings dam from where a portion is decanted and returned to the metallurgical process. The remainder is lost through seepage into groundwater or through evaporation. During these operations the water quality deteriorates as contaminant concentrations increase.

Mine closures may result in large waste rock dumps, mine tailings dams, discard dumps, old shafts or open adits, which become potential pollutant sources of receiving waters and aquifers. Mine waste water management studies(2) in the South African gold mining industry indicate the following:

- the gold mining industry uses 510 Mℓ/d of fresh make-up water, discharges about 440 Mℓ/d of effluent, primarily to evaporation dams, and circulates roughly 5500 Mℓ/d;

- the total amount of salt discharged from the gold mines is estimated at 1300 t/d;

- the relative increase in sulphate and total dissolved solids when passing service water through the stoping area is constant at about 20%;

- the use of carbonate neutralisation chemicals, as opposed to lime, results in mine waters that are significantly more stable;

The main uses of water in mines are:

- heat rejection (in underground condensers);
- cooling (of the underground environment, machines and various plants);
- mining (dust suppression, cleaning and pollutant transport)
- metallurgical processes

2. DIFFERENT TYPES/GROUPS OF CONTAMINANTS

For the purpose of this literature review the chemical contaminants will be discussed according to the following grouping and the focus will be on the gold and coal mining industry:

- heavy metals which include barium, boron, copper, iron, lithium, manganese, potassium, strontium, zinc (all water macro contaminants); antimony, arsenic, beryllium, bismuth, cadmium, chromium, cobalt, gold, lead, mercury, molybdenum, nickel, selenium, silver, tellurium, thallium, thorium, tin, titanium, tungsten, zirconium, vanadium, uranium (all water micro contaminants);

Properties of the following water contaminants are discussed in section 5

- Aluminium
- Calcium and magnesium
- Sodium
- Nitrogen contaminants, i.e. nitrates
- Sulphur contaminants, i.e. sulphates
- Chlorides
- Cyanide
- Radionuclides, e.g. uranium, thorium and radium
- Phosphorous group, i.e. phosphates

3. CONTAMINANT SOURCES

3.1 HEAVY METALS

Metals contained in the ore bodies may be oxidised, reduced, dissolved, absorbed or precipitated, depending on chemical conditions in the water. Under acidic conditions, metals may dissolve in water and be transported to a receiving stream.

Under neutral or basic conditions, metals adsorb onto organic materials, hydroxide precipitates and clay or precipitate in water, and are transported, resuspended and/or deposited in aquatic bottom sediments(1).

When water percolates through old workings at a slow rate it picks up substantial concentrations of salts. These streams are usually of a low volume but they may represent a significant salt load. The pH of the water tends to be very low and consequently very corrosive, the low pH is due to acid mine drainage caused by pyrite oxidation. Such waters cause metals such as iron and manganese and other heavy metals to become soluble in the mine service water.

The presence of iron and manganese in mine water is mainly attributed to the solution of minerals, i.e. as oxides and sulphides (e.g. pyrite). Dissolved iron and manganese tend to alter the appearance of the water, making it turbid yellow-brown to black. The precipitation of these metals may lead to problems in water treatment processes such as ion exchange, and anoxic limestone drains (ALD).

3.2 ALUMINIUM

Aluminium has only one oxidation state in aquatic systems, i.e. Al^{+3}. Oxidation and reduction processes which complicate Fe and Mn chemistry do not directly affect concentrations of dissolved aluminium. Concentrations of Al in mine water are primarily influenced by the solubility of $Al(OH)_3$. At pH levels between 5 and 8, $Al(OH)_3$ is highly insoluble and at pH values below 4, $Al(OH)_3$ is highly soluble (3).

The percolation of mine water through oxides or highly reduced environments has no effect on the concentration of Al, unless the pH also changes. In cases where the pH of mine water decreases (due to iron oxidation and hydrolysis), concentrations of Al increase because of the dissolution of aluminosilicate clays by the acidic water. When acidic mine water passes through anaerobic environments the increased pH resulting from carbonate dissolution or microbial activity causes the precipitation of $Al(OH)_3$.

3.3 CALCIUM AND MAGNESIUM

Calcium is a very common constituent in water and is dissolved from many types of minerals, especially limestone and gypsum. The calcium content of mine water is mainly due to the use of lime for neutralisation. High calcium concentrations are undesirable as they contribute to scaling in pipes and other equipment. SABS's recommended maximum permissible limit for calcium in drinking water is 200 mg/ℓ. Calcium is the major component of hardness in water.

Magnesium is a very common constituent in water and is dissolved from many different types of rock. The magnesium content in mine water is mainly due to dissolution from rock - specifically dolomite ($Ca/MgCO_3$). High magnesium concentrations are undesirable as they may contribute to the scaling of pipes and other equipment as a result of magnesium silicate or carbonate precipitation. At high pH values, in excess of 11,0, magnesium precipitates out as a light floc of magnesium hydroxide.

3.4 SODIUM

All sodium salts are highly soluble in water. The high chloride content of brine is usually associated with the sodium ion. Sodium is present in certain types of clay and feldspar.

Sodium is a very common constituent in water and is dissolved from many different types of rocks. The sodium content in mine water is mainly due to natural sources, for example, in the OFS fissure water has a high sodium chloride content, while treatment chemicals, for example NaOCl and Na_2CO_3 may also add sodium. The limit of sodium for potable water is 100 mg/ℓ.

3.5 NITRATE AND AMMONIA SPECIES

Nitrate is a common constituent in mine water, particularly in mines using nitrate-based explosives.

Nitrate, like ammonia, also enters water via the nitrogen cycle rather than through dissolving minerals. There is an increase of total nitrogen in sewage plant effluent at a range of 20 - 40 mg/ℓ as N above the level in water supply - a large proportion being ammonia with some nitrate.

Ammonia in mine water originates from the use of ammonium based explosives. Ammonia reacts with chlorine and can markedly impair its disinfection properties.

3.6 SULPHATES

Mine water is often contaminated with sulphate as a result of pyrite oxidation or dissolution in water from certain minerals, particularly gypsum. In waters with low pH values (typically below a pH of 4), oxidation of pyrite is enhanced by bacterial action. The bacterium *Thiobacillus ferrooxidans*, for example, thrive under low pH conditions. The high sulphate levels and calcium in many mines' water leads to calcium sulphate scaling and can cause problems in pipes, pumps, heat exchangers, cooling towers and desalination equipment. See literature review on acid mine drainage for more detail.

3.7 CYANIDE

Cyanide is used in the gold extraction process to dissolve gold from the ore body. It is essential to operate the process at a certain residual cyanide level to keep the gold recovery efficiency as high as possible. The metallurgical plant tailings are deposited on the tailings dam where the cyanide is destroyed.

Cyanide pollution of the environment occurs when plant spillage leaves the plant area and escapes catchment dams, or as seepage at the tailings dams.

Backfilling of underground workings is done with cycloned metallurgical plant tailings. The cyanide content in the backfill material is immobilised by the addition of ferrous sulphate before sending the material underground.

3.8 RADIONUCLIDES

Tailings from the extraction of gold and uranium still contain all the radioactive isotopes from the uranium, actinium and thorium decay series. Of particular concern for environmental protection is the amount of dissolved ^{226}Ra discharged with the tailings which has an extremely high radio toxicity. Analyses of tailings dams to assess the prevailing ^{226}Ra levels gave values of 15 to 85 pCi/g of ^{226}Ra for the Klerksdorp and OFS mining areas, where the richer uranium associated with gold is to be found, and 5 to 15 pCi/g on the East Rand as compared with Canada, USA and Australia where the ^{226}Ra content in tailings is measured in hundreds of pCi/g. Treatment of mine and tailings effluents by $BaCl_2$ to precipitate dissolved radium as (Ba, Ra) SO_4 as in the overseas mining countries is not practised in South Africa, since the radium concentrations in waste streams are too low to justify such treatment (4).

4. TREATMENT PROCESSES FOR REMOVAL OR PREVENTION OF MINE WATER CONTAMINATION

4.1 INTRODUCTION

Different treatment processes are applied for the removal of chemical contaminants. The following topics will be covered in more detail under the appropriate literature review.

- neutralisation
- flocculation and settling
- filtration - sand and multi-media
- cyanide
- radioactivity
- boilers and cooling water
- passive treatment in wetlands
- evaporation systems

4.2 REMOVAL OF HEAVY METALS

The total dissolved solids (TDS) content of mine water includes all dissolved ions. Desalination of mine water reduces the TDS value and the following advanced techniques for desalination have been investigated by various research bodies (7).

- Ion exchange which is feasible for treating acid mine water and for removal of chloride content for input to a metallurgical process (8).

- Seeded reverse osmosis which was evaluated on a pilot plant scale over 5000 operating hours. Results show high water recoveries (95 - 96%) with good salt rejection (10).

- Electrodialysis reversal shows promising results with successful desalination of hard brackish water of 4000 mg/ℓ TDS at water recoveries of up to 93% without pre-softening (11).

- Investigations on biological sulphate removal in an up flow packed bed reactor proved to be feasible as a means of treating high sulphate containing mine water with an average of 89% sulphate removal as well as calcium and heavy metal removal (12).
- An anaerobic filter process for treating high strength acid waste water under conditions of large amounts of heavy metals. 93% of COD removed was converted to CH_4, indicating low bacterial growth (13). This may also find application in the treatment of mine water.

4.3 REMOVAL OF ALUMINIUM

As already stated, at pH levels between 5 and 8, $Al(OH)_3$ precipitates, and the Al content in the water decreases. At a lower pH (2 - 3) the dissolved aluminium in the water may however be as high as 1200 mg/ℓ.

4.4 REMOVAL OF IRON AND MANGANESE

When oxidised in a pH range of 7 - 8,5, iron is almost completely insoluble and the actual residual iron after treatment is determined by how well the colloidal iron has coagulated and filtered from the water.

In an oxidised state, manganese is quite insoluble and can be lowered in concentration, even in an alum coagulation process by super-cholorination with adequate filtration, even at a pH as low as 6,5. The conventional process for removal of manganese is oxidation plus elevation of the pH to above 9, with retention of 30 minutes in a reaction vessel before filtration.

Organic materials can chelate manganese much as they can iron, so destruction of the organic matter is often a necessary part of the manganese removal process.

Because manganese accumulates in sediments, it is common to find high levels of manganese in deep water where none may be apparent at the surface.

4.5 REMOVAL OF MAGNESIUM

The magnesium hardness of a water is usually about 1/3rd of the total hardness with the remaining 2/3rds being calcium hardness.

Magnesium carbonate is more soluble than calcium carbonate and it is seldom a major component in scale. It must be removed along with calcium where soft water is required for boiler make-up or for process applications. It may be removed by lime softening and ion exchange.

4.6 REMOVAL OF CALCIUM

Calcium can be removed from water in softening operations and is a major factor in determining the Stability Index. Calcium reduction is often required in treating cooling tower make-up. Cation exchange methods can also be applied.

4.7 REMOVAL OF SODIUM

The only chemical process for removing sodium is desalination, using processes such as cation exchange, evaporation, reverse osmosis and electrodialysis reversal. Also see literature review on desalination.

4.8 REMOVAL OF NITRATES AND AMMONIA

A chemical process that removes nitrate is anion exchange, but nitrate can *be converted to nitrogen in a biological system by the action of the nitrifying* bacteria. Ammonia, ammonium, nitrate and nitrite are eventually reduced to nitrogen gas that is released. Ammonia can be removed by degasification, by cation exchange and by absorption by certain clays such as clinoptilolite.

4.9 REMOVAL OF CYANIDE

The destruction and removal of cyanide is discussed in the literature review on cyanide.

4.10 REMOVAL OF RADIONUCLIDES IN WATER

This subject is discussed in the literature review on radioactivity.

5. PROPERTIES OF WATER CONTAMINANTS

5.1 PROPERTIES OF WATER MACRO-CONTAMINANTS

5.1.1 Alkalinity

Alkalinity is a measure of the acid neutralising capacity of the water. It is mainly due to bicarbonate species. It has taste/physical implications and affects stability/corrosive potential of water. The alkalinity of natural water is rarely more than 500 mg/ℓ as $CaCO_3$.

5.1.2 Aluminium

Aluminium is the third most common element in the earth's crust, occurring in combination with silicates. Aluminium precipitates out of solution at neutral pH. Aluminium is classically regarded as non-toxic because of this insolubility. *It should be regarded as potentially toxic as it can go into solution in strongly* alkaline or acid conditions. Soluble aluminium has been implicated as a neurotoxin. With regard to irrigation water, aluminium can cause non-productivity in acid soils (pH < 5,5), but more alkaline soils at pH > 5,5 will precipitate the ion and eliminate any toxicity.

5.1.3 Ammonium

The ammonium ion NH_4^+ exists in dynamic equilibrium with free ammonia. The ammonium ion predominates at low pH, while the free ammonia predominates at high pH. The ammonium ion has low toxicity, while the free ammonia form has high toxicity, especially towards aquatic life. The criterion for ammonium therefore depends on the pH value.

5.1.4 Barium

Soluble barium is a muscle stimulant: the fatal dose for man is 0,8g as $BaCl_2$. It is toxic to the heart, blood vessels and nervous system and may cause kidney damage. Barium is readily precipitated from solution (and thus rendered non-toxic) by sulphate.

5.1.5 Bicarbonate

Bicarbonate is the main contributor to alkalinity in water.

5.1.6 Boron

The most important criterion with regard to boron is for irrigation. Boron is essential for plant growth, but the margin between beneficial amounts and toxic levels is narrow. The toxic level depends on the plant species. Boron in water can arise both naturally and from industrial or domestic effluents. It is not readily removed from solution, as boric acid and borates are readily soluble at neutral or alkaline pH.

5.1.7 Bromide

Bromide is important with regard to drinking water and for the protection of aquatic life.

5.1.8 Calcium

The calcium criteria for industrial uses are low, as calcium can cause scaling or precipitation of other chemicals. The calcium criteria for water consumption by living organisms are high, as calcium is non-toxic. Calcium is an essential element, being an integral part of bone (calcium phosphate). The human body normally needs up to 2g calcium per day. Calcium reduces the toxicity of heavy metals by hindering their adsorption. This is particularly noticeable in the decrease of the toxic potential of lead or zinc to fish, in the presence of high calcium concentrations.

5.1.9 Cerium

Cerium has low toxicity to man but high toxicity to aquatic life. Cerium occurs in association with the other so called lanthanide elements, and is the most common of the lanthanides

5.1.10 Chloride

Chloride usually occurs together with sodium in water. Chloride has low toxicity to those life-forms which have a mechanism for excreting excess chloride. To life-forms which do not have an efficient chloride regulating mechanism e.g. plants, chloride can be toxic. Therefore the median criterion for chloride in irrigation water is 100 mg/ℓ as compared to 1 500 mg/ℓ for livestock watering. Chloride is difficult to remove from water.

5.1.11 Copper

Copper is an essential element and only causes toxic symptoms in high concentrations. Water with a high copper content (above 1 mg/ℓ) has a disagreeable taste. Copper is, however, toxic to a number of plants at 0,1 to 1,0 mg/ℓ in nutrient solutions.

5.1.12 Detergents

Detergents cause aesthetic/physical problems e.g. foaming on water surfaces. They can also be toxic, particularly to aquatic life.

5.1.13 Fluoride

Fluoride is one of the elements which helps to prevent dental caries, but it has chronic long term toxicity in concentrations only slightly above the beneficial level. Fluoride concentrations in excess of 4 mg/ℓ may cause mottling of teeth, and in excess of 15 - 20 mg/ℓ may result in crippling skeletal fluorosis. Fluoride is fatal at around 4 - 5 g and toxic at around 250 - 450 mg. The desirable optimum fluoride concentration in drinking water depends on the average daily intake of water, and thus on the average daily maximum air temperature. For an average maximum air temperature of 16 °C, the optimum fluoride concentration is 1,0 mg/ℓ, decreasing to 0,7 mg/ℓ at 30 °C. Fluoride in irrigation water may be inactivated by neutral and alkaline soils.

5.1.14 Hardness

Hardness refers to the soap-neutralising power of a water. It is caused principally by calcium and magnesium ions which form insoluble stearates with soap. Hardness is undesirable for certain uses as it causes scaling. Hardness has been inversely correlated with the incidence of cardiovascular disease.

5.1.15 Hydrogen sulphide

Hydrogen sulphide is a foul smelling gas which characteristically smells like rotten eggs. Although toxic, it imparts unpleasant taste and smell to water at concentrations well below the toxic level. It is highly soluble in water.

5.1.16 Iodide

Iodide is essential for normal health, being necessary for the production of thyroid hormone. The margin between the essential nutritional requirement and toxic amounts is large.

5.1.17 Iron

Iron is the fourth most common element in the earth's crust. It is an essential element, being an integral part of the oxygen carrying red blood pigment, haemoglobin. The drinking water criteria for iron are based on aesthetic and not health considerations. Levels above 1 mg/ℓ impart a bitter-sweet astringent taste. Iron in water causes yellowish-brown discoloration. Iron is not toxic to plants in aerated soils, but can contribute to soil acidification and reduce availability of essential phosphorus and molybdenum.

5.1.18 Lithium

The lowest criterion for lithium is for protection of aquatic life. It is tolerated by most plants up to 5 mg/ℓ and is mobile in soil. It is toxic to citrus at low levels (>0,075 mg/ℓ). Lithium acts similarly to boron.

5.1.19 Magnesium

Magnesium is an essential element for human, animal and plant life and is non-toxic. The acceptable intake level for humans is 3,6 - 4,2 mg/kg/day. At high concentrations (400 mg/ℓ for sensitive people and 1 000 mg/ℓ for normal population) magnesium salts may have a laxative effect. For irrigation purposes, magnesium is a necessary plant nutrient as well as a necessary soil conditioner. Together with calcium it is responsible for water hardness.

5.1.20 Manganese

Manganese is an essential element for both plant and animal life. The limit for drinking water is based on aesthetic and not toxic considerations. Manganese, like iron, is a common cause of discoloured water. Prolonged intake of high manganese concentrations can, however, lead to serious neurotoxicity. Manganese is toxic to a number of crops at differing concentrations, but usually only in acid soils.

5.1.21 Nitrate & nitrite

Nitrite is unstable in water and is readily oxidised by bacterial action to nitrate.

These two elements can cause fatal methemoglobinemia in infants, and it is for this reason that the maximum permissible limit in drinking water has been set at 10 mg/ℓ. Sewage effluents and agricultural runoff are important sources of nitrate.

5.1.22 Phenols

The criterion for drinking water is based on the unpleasant taste of phenols. The fatal dose of phenol to man is estimated at 1,5 g. The limit for fish is low, both because of toxicity and also because of the phenolic taste that it imparts to fish. The toxicity of phenol-containing waste is often due to associated cyanide. In contrast to man and animals, phenol has a low toxicity to plant life.

5.1.23 Phosphates

Phosphorus is the 11 th most common element in the earth's crust, being found as phosphates. Phosphates are an essential nutritional requirement for all life forms. The higher animals and man require large quantities of phosphate, together with calcium, to form bone. Phosphates are not toxic. The main reasons for setting phosphate criteria are that phosphate is an algal nutrient (may result in eutrophication) and is indicative of pollution, e.g. from detergents, fertilisers, sewage etc.

5.1.24 Potassium

Potassium is the seventh most common element in the earth's crust. Potassium is an essential element for life, although it is toxic in large doses, as it disturbs the body's electrolyte balance.

5.1.25 Silica

Silica, i.e. SiO_2 or sand, is the basic constituent of the earth's crust. It tends to dissolve under alkaline conditions, but is non-toxic. The major problem associated with silica is of scaling where the water is used for steam generation

5.1.26 Sodium

Sodium is an essential constituent of the body fluids. Water containing high sodium levels is undesirable in patients with heart or kidney trouble. While sodium is required in limited amounts for most plant growth, high concentrations in the soil can be toxic and may, depending on the relative amounts of calcium and magnesium, decrease soil permeability. Sodium salts can be toxic to fish and animals at high levels and may affect osmoregulation in aquatic life.

5.1.27 Strontium

Natural strontium has little or no toxicity, in contrast to radioactive strontium-90 arising from nuclear fallout. Strontium is as common as fluorine in terms of abundance in the earth's crust.

5.1.28 Sulphate

Sulphate is non-toxic to animals and humans, except at very high levels when it exerts a purgative effect. In certain countries, water with sulphate levels in excess of 1 000 mg/ℓ is used for drinking without ill effects. For plant life, sulphate is slightly less toxic than chloride. The sulphate concentration must be considered where concrete is in contact with water, as sulphate promotes the corrosion of concrete.

5.1.29 Zinc

Zinc is moderately toxic to fish. Although zinc is non-toxic to humans, water containing high zinc concentrations should be screened for the highly toxic element cadmium, which is commonly associated with zinc. Zinc is toxic to many plants at widely varying concentrations, although its toxicity is reduced at pH > 6,0 and in fine textured or organic soils.

5.2 PROPERTIES OF WATER MICRO-CONTAMINANTS

5.2.1 Antimony

Antimony is chemically similar to arsenic and occurs primarily as an industrial pollutant. Antimony is a cardiac poison.

5.2.2 Arsenic

Arsenic is only moderately toxic, and has beneficial effects in small quantities. It was one of the first chemical agents shown to be carcinogenic. It is an accumulative body poison and causes neuropathy, vascular injury and skin cancer. Whether a given concentration of arsenic will be carcinogenic, toxic, non-toxic or even beneficial, depends very much on the chemical form in which it is present. Arsenic is an antoginist to selenium toxicity (selenosis) in animals. Its toxicity threshold to plants varies widely, ranging from 0,05 to 12 mg/ℓ.

5.2.3 Beryllium

Beryllium is chemically similar to aluminium. It varies in toxicity from highly toxic to moderately toxic. Its toxicity threshold in plants is species specific and varies from 0,5 to 5,0 mg/ℓ.

5.2.4 Bismuth

Bismuth is considered to be moderately toxic.

5.2.5 Cadmium

Cadmium is a highly toxic and accumulative poison to higher life forms. It is associated with cardio-vascular diseases, causes nausea and vomiting, and accumulates in the liver and kidneys. It is a recognised carcinogen and may cause sterility among males. At extreme levels it causes "Itai-Itai" disease, characterised by brittle bones and intense pain. Cadmium is also highly toxic to plants and also has the potential to accumulate in plants and soils to concentrations that may be harmful to humans. Cadmium is mainly an industrial pollutant and is often associated with zinc or lead. It is readily soluble in water and only precipitates at high pH values.

5.2.6 Chromium

Chromium is an essential element in human nutrition, without which insulin cannot function. Higher concentrations in certain forms, are toxic and cause skin disorders, liver damage and ulcers. There is also reason to believe that it may be carcinogenic in the chromate form. Chromium is not generally recognised as an essential growth element in plants. There is a lack of knowledge regarding its toxicity to plants and conservative limits are therefore specified for irrigation water.

5.2.7 Cobalt

Cobalt is an essential element in human nutrition, but is only needed in minute quantities. It exhibits a low toxic potential, except for irrigation where the toxic potential is moderate, depending on the plant species. Cobalt tends to be inactivated by neutral and alkaline soils.

5.2.8 Cyanide

Cyanide is more toxic to fish than to humans. It is rapidly degraded in the environment by bacterial action and is more toxic at low pH (when present as HCN), than at high pH. Free cyanide is more toxic than cyanide complexed to metallic elements such as iron.

5.2.9 Gold

Gold in a soluble form is highly toxic.

5.2.10 Lead

Lead is a highly toxic and accumulative body poison which affects nerve tissues, causing anaemia, brain damage and paralysis. In children, it may result in mental retardation with convulsions in later life. The toxicity of lead is inversely related to the calcium content of water, i.e., it is more toxic at low calcium concentrations than at high calcium concentrations. Lead can also inhibit plant cell growth at very high concentrations.

5.2.11 Mercury

Mercury is a highly toxic and cumulative poison, causing both brain and kidney damage. Mercury is concentrated in the aquatic food chain and transformed to the even more toxic methyl-mercury. The main danger posed by mercury pollution therefore results from eating fish contaminated by mercury. There is still much controversy regarding the levels of mercury which are dangerous, as its toxicity depends very much on other concomitant elements (i.e. the form in which the mercury is present). The essential element selenium, is a known mercury antagonist.

5.2.12 Molybdenum

Molybdenum in small quantities is an essential element for plants and animals. Both deficiency and excess cause disease symptoms. In animal nutrition, there is an interaction between molybdenum and copper. It is not toxic to plants at normal concentrations in soil and water, but it may be toxic to livestock if forage is grown in soils with high levels of available molybdenum.

5.2.13 Nickel

Nickel is widespread in the environment, being slightly more common than copper in the earth's crust. It is thought to be essential, in small quantities, for animal and plant life. It can be toxic to certain plants in concentrations exceeding 0,2 mg/ℓ. Its toxicity is reduced at neutral or alkaline pH.

5.2.14 Pesticides

Most organic pesticides have similar toxic effects and result in neurological and gastrointestinal symptoms. The toxicity of a pesticide when ingested by man, is not necessarily due to the pesticide itself, but may arise either from some other highly toxic impurity present as a trace in the pesticide, or from transformation of the pesticide to a more toxic metabolite within the body.

5.2.15 Radioactivity (α+β)

The criteria for radioactivity are based on the assumption that long term exposure to low levels of radioactivity poses a cancer risk. The danger posed by radiation depends very much on the type (α, β, γ) of the radiation as well as on the energy of the radiation. Besides the well known natural radioactive elements uranium, thorium and radium, a number of other natural elements are very slightly radioactive, viz. potassium, rubidium, certain lanthanide elements, zirconium, hafnium, vanadium, tantalum, rhenium, platinum and tellurium. Of these elements, potassium and vanadium are essential nutritional elements.

5.2.16 Radium

Radium is the most radio toxic of all the naturally occurring elements. Its isotope of mass 226 is the most hazardous. Radium is bone-seeking, has a long half-life (1 600 years), and as an alpha-emitter, has a high potential for biological damage when taken up in the human body.

5.2.17 Selenium

Selenium is a rare element. It is essential in very small quantities for normal health, serving as an anti-oxidant and involved in vitamin E metabolism. The margin between beneficial levels and toxic levels is narrow. Both absence and excess quantities of selenium result in disease. It is also toxic to plants at concentrations as low as 0,025 mg/ℓ, and toxic to livestock if forage is grown in soils with relatively high levels of added selenium.

5.2.18 Silver

Silver is only moderately toxic to man and its chief toxicity is towards aquatic life. Silver is rapidly immobilised in the environment as insoluble silver sulphide.

5.2.19 Tellurium

Tellurium is a rare element (similar abundance to gold) and is chemically similar to selenium.

5.2.20 Thallium

Thallium has about the same abundance in the earth's crust as mercury. It is similar in toxic potential to arsenic, and thallium sulphate has been used as rat poison. The danger of thallium pollution is that thallium tends to remain in solution, even at high pH values. It is not easily removed from water.

5.2.21 Thorium

Thorium is a natural radioactive element of similar abundance in the earth's crust as molybdenum. Its toxicity is due to its radioactive nature as an alpha-emitter.

5.2.22 Tin

Tin is an essential element in animal nutrition and has a moderate to low toxicity, as it tends to precipitate out of solution and is not readily absorbed. Tin is effectively excluded by plants and the specific tolerance of plants therefore is not known.

5.2.23 Titanium

Titanium is the ninth most common element in the earth's crust. Titanium does not usually occur in water in high concentration because its oxide is highly insoluble. Its effect on plants is similar to that for tin.

5.2.24 Tungsten

Tungsten is similar in abundance to copper in the earth's crust. Little is known about the health effects of this element. Tungsten also has the same effects on plants as tin.

5.2.25 Uranium

Uranium is a natural radioactive element, and is often found in ground water. The general feeling is that uranium is not as hazardous as might be expected for a radioactive element, although its decay products could be highly toxic. Recent information, mainly from the USA, indicates that it may be more carcinogenic than previously thought. For soluble uranium ingested by humans, the absorption is 0,5 to 5 per cent. Uranium may be chemically toxic to the kidneys.

5.2.26 Vanadium

Vanadium is suspected of being an essential element in human metabolism, and is known to have an effect on cholesterol metabolism. Vanadium is toxic in excess. Vanadium is an essential nutrient for the growth of plants, but is also toxic to many plants at relatively low concentrations (5).

6. CONCLUSION

Chemical contaminants in mine water systems are complex with synergistic effects in the environment. Two major sources of contamination do seem to prevail. Pyrite oxidation leads to acidic, saline conditions with mobilised (dissolved metals) and nutrient enrichment from explosives and sewage leads to eutriphication, pH fluctuations and decreased oxygen contents.

Neither source of pollution can be cost-effectively eliminated from mine operations and both need to be controlled and properly managed.

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MICROBIOLOGICAL CONTAMINANTS

1. INTRODUCTION

Organisms in water that are undesirable for a specific use of the water can be seen as microbiological contaminants. This literature review should also be read in conjunction with the literature reviews on disinfection, water use standards and potable water treatment.

There are two major sources of microbiological contamination in mine water circuits; direct faecal pollution of water being circulated through the stope areas and contamination of droplets from spray ponds and bulk air coolers in the fridge plants (see literature review on boilers and cooling). The water spray cooled contaminated air is then used for ventilation and species like *Legionella* cause infection of workers through inhalation. Microbiological contaminants from faecal pollution may be ingested through drinking of service water that is not disinfected. The chilled service water is often preferred above provided potable water, and therefore needs to be disinfected.

The organisms that are responsible for microbiological contamination, will be discussed in two sections; pathogens and those affecting industrial processes.

2. PATHOGENS

Freshwater provides an important route of exposure to water-associated pathogens. A wide range of infectious agents and their associated diseases can be transmitted by ingestion, inhalation or skin contact. Diseases like cholera, typhoid fever, salmonellosis, bacillary dysentery, viral gastro-enteritis and hepatitis A may be incurred.

Because there are numerous water borne pathogens that may cause potential health effects in mine service water and water that may be released by mines, the focus will fall on the most important organisms and the organisms used as indicator organisms.

2.1 FAECAL COLIFORMS/*ESCHERICHIA COLI*

Total coliform bacteria are frequently used to assess the general hygienic quality of water, while faecal coliforms indicate faecal pollution. The presence of *E. coli* is used to confirm the presence of faecal pollution by warm-blooded animals. While it is recognised that *E. coli* is the preferred indicator of faecal pollution and health risk, it is probable that many smaller water laboratories in South Africa may have facilities for analysis of total or faecal coliforms but not for confirmation of *E.coli* this makes the total coliform group a better indicator organism. *E. coli* usually comprises approximately

97% of coliform bacteria in human faeces but some faecal coliform tests may enumerate *Klebsiella* spp. and a few other bacterial strains, which originate from non-faecal origin. Hence *E. coli* is the preferred indicator of faecal pollution of natural water (4).

The term faecal coliforms (alternatively thermotolerant coliforms or presumptive *E.coli*, refers to coliform bacteria capable of growth on selective media at elevated temperature, generally 44,5 °C. Probably the most common method of detection is membrane filtration of water samples, and incubation of membranes on m-FC agar. Faecal coliforms are those bacteria which, under these conditions produce blue colonies within 20-24 hours of incubation. *E. coli* are detected as faecal coliforms which test indole positive at 44,5 °C.

Both *E. coli* and Faecal coliforms are generally expressed as health threat, but indicate the presence of other pathogens of which some are described in Table 1(from Pulles (1993)).

Together with coliforms and *E. coli*, faecal streptococci may be used to differentiate between human faecal pollution and animal faecal pollution.

Faecal streptococci are one of the bacterial types found in faeces. The ratio of faecal streptococci (Fs) to faecal coliform bacteria (Fc) is indicative of the origin of the faecal pollution. A Fc/Fs ratio of > 4,0 is indicative of human faecal pollution, while a Fc/Fs ratio of < 0,6 is indicative of animal faecal pollution (1).

2.2 LEGIONELLA

Legionella spp. are aerobic bacteria which can cause pneumonia, diarrhoea and nausea and are transmitted to man by inhalation of aerosols into the lungs. *Legionella spp.* are found in cooling towers, evaporative condensers and potable water distribution systems, with optimal growth at 35-37 °C in humidified air. They can survive in a wide range of conditions in water : pH 5,5-8,1; temperature 5-63 °C; and dissolved oxygen 0,3-9,6 mg/ℓ. They are relatively chlorine tolerant and require a higher than normal chlorine residual of 4-6 mg/ℓ for disinfection(1).

Although heat rejection devices (cooling towers and evaporative condensers) have been linked to outbreaks of legionnaires' disease, recent evidence suggests that potable water distribution systems are the primary reservoirs of *Legionella pneumophila*. Inhalation of aerosols containing the organism, instillation of the organism into the lung via medical manoeuvres of the respiratory tract, and aspiration of contaminated water into the lung are *the most likely modes of transmission and infection*(15).

The major manifestation of Legionnaire's disease is pneumonia: Patients typically have fever and cough, usually feel lethargic, and can become disoriented. Unlike patients with other pneumonia's, patients with Legionnaires' disease often have severe gastrointestinal symptoms, including diarrhoea, nausea and vomiting. Pontiac fever is the second clinical syndrome associated with Legionella infections. With this syndrome, patients do not have any respiratory symptoms, but have headache, muscle aches and malaise. Patients often recover completely without any treatment.

Studies to investigate the survival and multiplication of *Legionella spp.* in public drinking water supplies, have shown that residual chlorine in treated water supplies inhibits the regrowth and distribution of *Legionella spp.* effectively(16).

TABLE 1 : Bacteria of Health Significance in Water(1)

Genus	**Species**	**Host(s)**	**Disease**	**Transmission**
Salmonella	Several hundred serotypes known to be pathogenic to man. Eg. *S. typhi,* *S. enteritides,* *S. typhimurium*	Found in human and animal guts, polluted water, contaminated foods	Typhoid fever from *S. typhi,* salmonellosis from other species; both cause acute diarrhea and cramping; typhoid fever can be fatal	Transmitted through water or food processed with contaminated water
Shigella	Four species cause disease in man: *S. sonnei, S. flexneri,* *S. boydii* *S. dysenteriae*	Rarely found in hosts other than man	Shigellosis, or bacillary dysentery, causes fever and bloody diarrhea	Transmitted via contaminated food, polluted water, and person-to-person contact
Leptospira	Common human isolates include *L. pomona,* *L. autumnalis,* *L. australis*	Found in infected humans and animals. Excreted via urine	Leptospirosis: acute infections of kidney, liver, central nervous system	Transmitted through skin abrasions or mucous membranes to blood stream; may come from contact with animal carriers and/or polluted water
Vibrio	*V. cholerae*	Humans	Cholera: an acute intestinal disease causing vomiting, diarrhea, dehydration, can be fatal	Contaminated water, person-to-person contact
Entero-pathogenic *E. coli*	Various serotypes	Warm-blooded animals	Diarrhea, urinary infections	Sewage contaminated water or food; person-to-person via physical contact
Yersinia	*Y. enterocolitica,* *Y. pseudo-tubercolosis*	Birds and mammals	Diarrhea, fever, vomiting, anorexia, acute abdominal pain, abcesses, septicemia	Animal-to-man, person-to-person, contaminated water
Myco-bacterium	*M. tuberculosis,* *M. balnei, M. bovis*	Man, diseased cattle	Pulmonary or skin tuberculosis	Typically airborne, but sewage-contaminated water is a demonstrated route

Assessments of inactivation of *Legionella pneumophila* by organic halamine, free chlorine and a mixture of the organic halamine and free chlorine, indicated that the organic halamine was found to have superior stability in solution and to exhibit adequate disinfectant potential over a period of 1 month of repeated reinoculations of fresh bacteria. The combined halamine exhibited great potential for use in maintaining closed cycle cooling water systems free of *Legionella pneumophila*(17).

2.3 VIRUSES

Viruses have been associated with water borne diseases such as gastro-enteritis, hepatitis and respiratory illness(7). Coliphages are bacterial viruses which have been suggested as indicators of the presence and fate of human viruses in water. They infect and replicate in *E.coli*, and may infect related coliform bacteria. Since Coliphages require these bacterial hosts for replication, the presence of coliphages probably also indicate their presence(8).

Coliphages may be subdivided into somatic coliphages, which infect *E. coli* by absorbing to viral receptors in the cell wall, and male specific coliphages which infect *E. coli* by specifically adsorbing to threadlike appendages (4). Somatic phages may multiply in the aquatic environment by infecting bacteria other than *E. coli* (8).

Coliphages are detected in a plaque assay, using an *E. coli* C strain as host. Viral defence mechanism in this strain have been disabled to allow infection by a wide range of coliphages. Coliphages are often expressed as plaque forming units (pfu) or counts per 10 mℓ. Often counts per 100 mℓ are used to facilitate comparison with analyses of other indicator organisms, particularly faecal coliforms/*E. coli*.

Enteric viruses refer to viruses which are primarily transmitted by faecal-oral route, i.e. which are transmitted by faecally contaminated water. This group includes enteroviruses, adenoviruses, reoviruses, rotaviruses, hepatitis A and Norwalk virus. Possible health effects associated with the presence of such viruses in water include paralysis, meningitis, hepatitis, respiratory illness and diarrhoea. Levels of enteric viruses may be expressed as $TCID_{50}$ per 10 mℓ, referring to the dose of virus which required to cause 50% infection in tissue culture (4).

Detection methods for enteric viruses are expensive and labour intensive, while coliphages are detected easier, rapidly and cost-effectively. For this reason coliphages are used as indicator organism for pathogenic viruses described in Table 2.

TABLE 2 : Viruses of Health Significance in Water(1)

Group	No.types	Disease
Adenoviruses	37	Respiratory illness, conjunctivitis
Reoviruses	3	Respiratory disease, diarrhoea
Rotaviruses	> 3	Infantile gastro-enteritis
Enteroviruses		
Polioviruses	3	Poliomyelitis
Coxsackie viruses A	23	Aseptic meningitis
Coxsackie viruses B	6	Aseptic meningitis
Echoviruses	31	Aseptic meningitis, cold
Other enteroviruses	> 4	AHC, encephalitis
Hepatitis A virus	1	Infectious hepatitis
Norwalk and related gastrointestinal viruses	> 3	Gastro-enteritis

2.4 PROTOZOAN PARASITES

Comparison of the occurrence of indicator organisms and protozoan parasite cysts and oocysts in various aquatic environments, including South Africa, show that common indicators can't be used reliably to indicate parasite occurrence and fate (9, 10, 11).

TABLE 3 : Protozoans of Health Significance in Water(1)

Protozoan	Host(s)	Disease	Transmission
Acanthamoeba castellani	Fresh water, sewage, man	Amoebic meningoencephalitis	Gains entry through brasions, ulcers, and as secondary invader during other infections
Balantidium coli	Pigs, humans	Balantidiasis (dysentery)	Contaminated water (pig faeces)
Entamoeba histolytica	Humans	Amoebic dysentery	Contaminated water
Giardia lamblia	Animals, humans	Giardiasis (gastroenteritis)	Contaminated water
Nagleria fowleri	Soil, water, decaying vegetation	Primary amoebic meningoencephalitis	Nasal inhalation with subsequent penetration of nasopharynx; exposure from swimming in freshwater lakes

Infection by protozoan parasites, occurs by ingestion of cysts or oocysts and disease takes the form of gastro-enteritis, diarrhoea, vomiting and anorexia. Acute, chronic and asymptomatic manifestations of infection are known (see Table 3). Monitoring of protozoan parasites is difficult and no standard monitoring practices, concentration procedures or detection methods have been developed as yet (4).

Standard staining techniques may be employed, but the most commonly used methods for detection of *Giardia* and *Cryptosporidium* (*Cryptosporidium parvum* has been recognised as a waterborne human pathogen) utilise fluorescent conjugated antibody staining. Levels are often expressed as number of cysts or oocysts per 10ℓ, and if possible a distinction should be made between viable and non viable cysts (4).

2.5 HELMINTHS OF HEALTH SIGNIFICANCE IN WATER

This group of nematodes and worm like organisms, are parasites that have an aquatic dwelling phase in their life cycle, usually in the form of eggs or cysts(12). The eggs or cysts ingested from contaminated water have an extremely low infective dose, theoretically one cyst or oocyst will be sufficient to cause infection (4).

Transmission, host and diseases caused by the most common helminths are summarised in Table 4.

TABLE 4 : Helminths of Health Significance in Water(1)

Agent	Host(s)	Disease	Transmission
Ascaris lumbricoides (intestinal roundworm)	Man	Ascariasis - moderate infections cause digestive and nutritional problems, abdominal pain and vomiting; live worms passed in stools or vomited; serious cases involving liver can cause death	Ingestion of infected eggs from soil, salads and vegetables contaminated with eggs from human faeces
Schistosoma mansoni, S. haematobium, S. intercalatum, S. japonicum	Man, domestic animals and rats serve as primary hosts; snails act as necessary intermediate host	Schistosomiasis - a debilitating infection where worms inhabit veins of host; chronic infection affects liver or urinary system	Water infected with cercariae (larvae) which have developed in snails; penetration through human skin; eggs excreted via urine or faeces and miricidium (larvae) develop in water and re infect snails
Necator americanus or Ancylostoma duodenale (hookworm)	Man	Ancylostomiasis - hookworm disease; debilitating disease associated with anaemia; heavy infestations can result in retardation; light infections produce few effects	Eggs from deposited faeces develop into larvae which penetrate the skin; ancylostoma can be acquired orally
Strongyloides stercoralis	Man and possibly dogs	Strongyloidiatis - intestinal infection causing cramps, nausea, weight loss, vomiting, and weakness; rarely results in death	Ineffective larvae in moist soils resulting from faecal contamination penetrate skin and reach digestive system via venous and respiratory systems
Trichuris trichiura (whipworm)	Man	Trichuriasis - a nematode infection of the large intestine; often without symptoms, but heavy infestations result in abdominal pain, weight loss and diarrhoea	Ingestion of eggs in soil and/or vegetables contaminated with faecal material

3. MICROBIOLOGICAL CONTAMINANTS THAT MAY AFFECT INDUSTRIAL PROCESSES

3.1 POTENTIAL EFFECTS OF BACTERIA ON INDUSTRIAL PROCESSES

Bacteria, the largest group of troublesome organisms (as summarised in Table 5), cause the most varied problems. They are usually classified in water treatment by the types of problems they cause: slime forming bacteria, iron depositors, sulphate reducers and nitrifying bacteria.

Many bacteria secrete a mucilaginous substance that encapsulates the cell, shielding it from direct contact with water, so the cell is protected from simple toxic biocides. Control of encapsulated bacteria usually requires both oxidation and dispersion of the protective sheath so that the biocide can reach the cell.

Aerobic bacteria require oxygen, and are often found in cooling towers and open systems. Anaerobic bacteria obtain their energy from reactions other than oxidation of organic substances, the reduction of sulphate to sulphide is an example. Sulphides are corrosive to most metals used in water systems, including mild steel, stainless steel and aluminium.

Iron depositors convert ferrous iron in water to insoluble Ferro hydroxides that deposit and accelerate corrosion rates, which produces additional soluble iron. The cycle accelerates and can clog the whole system with iron deposits(2).

TABLE 5 : Some General Of Microorganisms That Affect The Quality Of Water In Distribution Systems(1)

General	**Possible effect**
Actinomyces	Tastes and odours
Arthrobacter	Colour production (porphyrins): Slime formation
Bacillus	Nitrate reduction; corrosion; potential indicator antagonist
Beggiatoa	"Red" water, sulphur oxidation
Crenothrix	"Red" water (iron bacteria)
Desulfovibrio	Corrosion; hydrogen sulphide production ("black water")
Gallionella	"Red" water (iron bacteria)
Leptothrix	"Red" water (iron bacteria)
Methylmonas (Methanomonas)	Methane oxidation
Micrococcus	Nitrate reduction; corrosion; potential indicator antagonist
Nitrobacter	*Nitrate production; possible corrosion and slime formation*
Nitrosomonas	Nitrite production; possible corrosion and slime formation
Sphaerotilus	"Red" water
Streptomyces	Taste and odour
Thiobacillus	Acid production (pyrite oxidation), corrosion, slime production

3.2 POTENTIAL EFFECTS OF FUNGI ON INDUSTRIAL PROCESSES

Saprophytic fungi spores may be introduced to supporting wood structures underground, where the wood can be destroyed over time(2). As saprophytic fungi prefer dark, moist and warm surroundings, this potential contaminant can flourish underground if wood used in stacks, is not disinfected and treated with fungicide.

There are not many other water contaminating fungi types that pose a specific threat to mining operations.

3.3 POTENTIAL EFFECTS OF ALGAE ON INDUSTRIAL PROCESSES

Algae are ubiquitous in aquatic environments and are a vital part of water bodies, providing a source of food and energy for fish and other aquatic organisms.

Excess algae or undesirable algal types can, however, become a nuisance and interfere with the desirable uses of water. Attached filamentous or colonial algae may become detached and become a hazard to industrial pumps and machinery. Algal blooms (eutrophication) may have the same effect and can cause water to become toxic and cause skin irritations (4).

Eutrophication may be induced more easily in mine waters than other waters, because of increased water temperature from geothermal heating and increased nitrate concentrations from explosives. The heavy algal growth and resultant collapse of the algal populations lead to pH fluctuations and eventually oxygen depletion, which affects other chemical contaminants through redox potential.

In general, guidelines are based on limiting nutrient (nitrogen and phosphorous) concentrations, thus limiting algal growth (4).

4. CONTROL OF MICROBIAL ACTIVITY AFFECTING INDUSTRIAL PROCESSES, AS OPPOSED TO TOTAL DISINFECTION

All water treatment processes are affected by the presence of microbes(2). Many redox reactions are biologically mediated. In most cases, microbial affects are detrimental to the water-using process or system.

Disinfection of drinking water and sterilisation of food processing and hospital equipment are examples where the goal is to kill all microbes, a complete kill is however costly and not always necessary. Cooling water utilities, steel mills, mines and other industrial plants are treated to control microbe populations at levels tolerable to the system without complete sterilisation.

In the mining industry there is one specific organism, Thiobacillus ferrooxidans, that causes pyrite oxidation which in turn results in sulphuric acid production, increased sulphate concentration and mobilised (solubilised) heavy metals. See separate literature review on acid mine drainage.

Planning an effective microbial control program for a specific water treatment process requires an examination of:

1. The types of organisms present in the water system and the associated problems they can cause.

2. The population of each type of organism that may be tolerated before causing a significant problem

Also see literature reviews on boilers and cooling.

5. SUMMARY

Microbiological organisms, and specifically bacteria and viruses, cost the mining industry large amounts of money through necessary disinfection, scaling, corrosion and fouling. Unlike complete disinfection of pathogens, most organisms need to be controlled, as it will not be cost effective to eliminate the organisms in water systems completely.

One specific organism relating to mine water problems, is *Thiobacillus*, that causes acid mine drainage (see literature review), that increases the sulphate concentrations and salinity, and also mobilises metals in the water. Research in the field of controlling pyrite oxidation by bacteria and using sulphate reducing bacteria in passive treatment systems to reduce the sulphate concentrations in the mine water, is underway (13, 14).

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NEUTRALISATION

1. INTRODUCTION

Neutralisation is the process whereby the pH of acid or alkaline water is corrected, usually by chemical addition, towards a neutral pH of 7. The phenomenon and mechanisms of acid mine drainage (AMD) are well documented and the subject is discussed in a separate literature review, but the occurrence of acid water is one of the biggest water management problems in the mining industry. Neutralisation may be applied to this water for any of the following purposes:

- treatment of effluent prior to its discharge to the environment.
- adjustment and control of pH as pretreatment for a downstream unit process, such as flocculation for underground settling.
- pH and carbonate correction to reduce scaling or corrosion characteristics.

The chemistry of neutralisation is complex but an elementary understanding is essential if the various neutralisation techniques are to be understood.

2. CHEMISTRY AND RELATED CONCEPTS

In order to understand the process of neutralisation it is useful to explain the chemistry and theory of pH. Although the control of scale and corrosion is only one objective of neutralisation, the concepts are included in this section as an understanding of the chemistry is essential in implementing successful treatment processes. The concepts are also discussed, at various levels of detail, in the numerous texts which are available on the subject(1),(2),(3).

2.1 pH CHEMISTRY

Pure water consists of H_2O molecules as well as the dissociated ions, H^+ (hydrogen ion) and OH^- (hydroxide) which result from the reversible reaction

$H_2O \longleftrightarrow H^+ + OH^-$ (1)

When a reversible reaction is at equilibrium the rates of the forward and reverse reactions are equal and for the example of water dissociation (1), the following expressions can be written.

Rate of forward reaction = $k_1[H^+][OH^-]$ (2)
Rate of reverse reaction = $k_2 [H_2O]$ (3)

where k_1 and k_2 are rate constants. At equilibrium, and according to the Law of Mass Action, equations (2) and (3) become

$[H^+][OH^-] = k_2/k_1 = K$ (4)

where K is known as the thermodynamic dissociation constant. This factor can also account for the concentration of water molecules, $[H_2O]$ which can be considered as constant, due to the fact that the dissociation of water is weak. The value of K at 23 °C is 1×10^{-14} mole litre^{-1} and in a neutral water this implies that

$[H^+] = [OH^-] = 10^{-7}$ mole litre^{-1}

From the definition of pH as pH = $-\log_{10}[H^+]$ it follows that a neutral water will have a pH of 7 and an acidic water, with a high concentration of H^+ ions, will have a pH < 7. This is, however, a simplified interpretation of neutrality as there are other complicating factors which are discussed below.

2.2 CARBONATE CHEMISTRY

Water contains various dissolved species and ions, the most important of which, from a neutralisation aspect, include CO_2, H_2CO_3, HCO_3^-, CO_3^{2-}, Ca^{2+}, $CaCO_3$, H^+ and OH^-. These species comprise the carbonate system, the intricate balance of which is governed by complex chemistry, beyond the scope of this review. It is necessary however, to include a discussion on the role of carbon dioxide, CO_2 and bicarbonate, HCO_3^- in particular, in order to gain a full understanding of acidity and alkalinity.

Carbon dioxide is present in the atmosphere and is a product of fuel combustion processes and respiration. The concentration in soil is much greater as a result of soil organism respiration. Dissolved CO_2 reacts with water to form the weak acid, carbonic acid, which then dissociates into the hydrogen and bicarbonate ions according to reaction (5).

$CO_2 + H_2O <\!\!-\!\!-\!\!-\!\!> H_2CO_3 <\!\!-\!\!-\!\!> H^+ + HCO_3^-$ (5)

This water is able to dissolve magnesium and calcium from dolomite, which is a combination of calcium carbonate and magnesium carbonate as shown below.

$CO_2 + H_2O + MgCO_3 \dashrightarrow Mg(HCO_3)_2 <\!\!-\!\!-\!\!-\!\!> Mg^{+2} + 2(HCO_3^-)$ (6)

$CO_2 + H_2O + CaCO_3 \dashrightarrow Ca(HCO_3)_2 <\!\!-\!\!-\!\!-\!\!> Ca^{+2} + 2(HCO_3^-)$ (7)

The calcium and magnesium ions are responsible for the hardness of the water and the bicarbonate ions, along with carbonate ions, CO_3^{-2}, give the water most of it's alkalinity.

2.3 ALKALINITY AND ACIDITY

Alkalinity is the acid neutralising capability of a water. It is equal to the amount of standard strong acid which would be required to take the water to a defined end point. This end point may be the methyl-orange end point, which is approximately pH 4,5 depending on the initial condition of the water, and this alkalinity is then termed M alkalinity. P alkalinity is determined by titration to the phenolphthalein endpoint of pH 8,4. The relationship between these two values gives an indication as to the quantities of alkaline hydroxides, carbonates and hydrogen carbonates in the sample according to Table 1.

Table 1 : Contributions of various ions to alkalinity.

Dissolved Salts	P Alk = 0	P Alk < M Alk/2	P Alk = M Alk/2	P Alk > M Alk/2	P Alk = M Alk
OH^-	0	0	0	2 P Alk - M Alk	M Alk
CO_3^{2-}	0	2 P Alk	M Alk	2(M Alk - P Alk)	0
HCO_3^-	M Alk	M Alk - 2 P Alk	0	0	0

The buffering capacity is closely related to the amount of alkalinity in the water where buffer capacity is the ability of the water to absorb acid without a substantial decrease in pH.

Acidity is defined as the base neutralising capacity of a water and is determined by the following expression:

$$\text{Acidity} = CO_2 + H_2CO_3 + HCO_3^- + H^+ - OH^- \qquad (8)$$

where these terms are usually expressed in mg/ℓ as $CaCO_3$. This end point is more difficult to determine and acidity is thus seldom measured.

2.4 SCALE AND CORROSION

The term saturation pH which is denoted as pH_S, refers to that pH value of a water in which the product $[Ca^{+2}][CO_3^{2-}]$ is equal to the solubility product constant for the precipitation reaction of $CaCO_3$. In the over saturated condition, there is a tendency for the precipitation to form and the water is known as scale forming or protective. In this condition the distribution of CO_2 within the water is such that the concentration of free CO_2 is lower than that required at equilibrium. The formation of scale is prevented by treatment which is aimed at decreasing the pH, softening or removal of carbonates.

Conversely, at a pH value less than pH_S, the water tends to be corrosive and is known as an aggressive water. The low pH is due to an excess of CO_2, which is in excess of that required for the saturated condition. Neutralisation treatment aims either to remove or transform this excess CO_2.

There are several graphical methods available to determine whether a water is aggressive or scale-forming, the simplest of which is the Langelier diagram which accounts for temperature, ionic strength and alkalinity concentrations. Langelier also established the Langelier Saturation Index (LSI) which is simply the difference between the actual pH and the saturation pH, denoted pH_S.

$$LSI = pH - pH_s \quad (9)$$

From the preceding discussion, it is can be deduced that a negative LSI implies a condition in which any calcium carbonate, which has been precipitated, is stripped away. A positive LSI implies a scale-forming, protective condition. Under certain conditions this may be an over-simplification and in 1944 Ryznar proposed a new index known as the Ryznar Stability Index (RSI).

$$RSI = 2pH_S - pH \quad (10)$$

where the following relationships hold:
RSI = 6,0 (water stable)
RSI > 7,5 (corrosion is marked)
RSI > 9,0 (corrosion may be serious)
RSI < 6,0 (scale forming condition)

The determination of pH_S is complex and various simplifying methods of calculation and interpretation of calcium carbonate saturation[4, 5, 6] exist. Computer programs such as Stasoft and Aquachem have also been developed to facilitate these calculations.

3. OBJECTIVES OF NEUTRALISATION

Neutralisation treatment is applied with the objective of correcting pH, either as pretreatment for a downstream process, or prior to final discharge to the receiving environment, or as a stabilisation process for scaling or corrosive waters.

3.1 pH CORRECTION FOR GENERAL USE OR DISCHARGE

According to the general effluent standard, the pH of highly alkaline or acidic waters needs to be corrected before it is discharged to the environment. The standards are fairly lenient in terms of pH and the permissible range is between 5,5 and 9,5. If the water is intended for other use, then different standards apply. In terms of mining applications, these ranges are specified in the COMRO guidelines for various end-uses. These values are summarised in Table 2 below.

Table 2 : Recommended pH limits for various applications

Application or end use	Recommended pH range
SABS Drinking water	6 - 9
General Effluent Standard	5,5 - 9,5
COMRO (Mine service water)	5,5 - 8,5
COMRO (Hydropower - carbon)	6 - 8,5
COMRO (Hydropower - galvanised	7 - 8,5
COMRO (Cooling water -copper)	5,5 - 8,5
COMRO (Cooling water - stainless)	3,0 - 8,5
COMRO (Cooling water - titanium)	1,0 - 8,5
Steam generation	7 - 10
Irrigation	4,5 - 9
Dams - protection of aquatic life	6 - 9

3.2 pH CORRECTION AS PRETREATMENT

Many processes such as coagulation, flocculation, sedimentation and filtration are directly or indirectly affected by the pH of the influent water. In particular, the upstream flocculant dosing of underground settling and surface thickening is susceptible to pH. Effective control thereof is essential in reducing both chemical dosing costs as well as the indirect costs due to abrasion, through high suspended solids contents. Flocculants, particularly the commercial polymer varieties, are expensive and the payback time of an effective pH control system can be very short. The control is however, dependant on which neutralisation process is used and particular problems are discussed in the relevant section. The effect of neutralisation on floc generation is discussed in the literature(7) and the subject of flocculation and settling is discussed more fully in a separate literature review.

Neutralisation is necessary as pretreatment for the oxidation and precipitation processes in acid mine drainage (AMD) treatment. The US Bureau of Mines have developed the In-line Aeration and Neutralisation System (ILS) which treats AMD in a pipe and involves no moving parts(8).

3.3 STABILISATION OF SCALING OR CORROSIVE WATERS

The conditioning of waters which are scaling or corrosive is perhaps the most important of neutralisation objectives in terms of possible cost savings. The problems of scale formation in the mining industry are well studied(9),(10) and the direct costs of scaling type waters are due to the increased pumping costs as well as increased maintenance required for the periodic removal of scale and pipe cleaning. However, these costs are negligible in comparison to the replacement costs of pipelines, vessels and pumps which are subjected to corrosive conditions. These problems are well documented for both the mining industry(11, 12, 13, 14) as well as other sectors(15, 16). It is clear from the brief chemistry background given in Section 2 that the scaling and corrosion tendencies of a water are dependant on the carbonate balance, which is pH dominated. The choice of neutralisation process is therefore not as straightforward as that for simple pH correction treatment.

4. METHODS OF NEUTRALISATION

There are both chemical and physical methods of adjusting the carbonate balance and achieving a more stable water.

4.1 CHEMICAL METHODS

Neutralisation can be achieved through the addition of alkaline or alkaline-earth based reagents. Some of the properties, advantages and disadvantages of the three most common neutralisation chemicals are summarised in Table 3. A fuller discussion of the neutralising capacities of sodium and calcium based products is given in the literature(17). These and other properties are discussed below.

4.1.1 Lime

Lime may be bought in a variety of forms depending on the quantities required and local availability. Commercial, hydrated lime $(Ca(OH)_2)$ is expensive and will only be used where small quantities are required. Usually, it is bought in the pebble form called quicklime (CaO) which may be railed or trucked to the mine and stored in large silos. This quicklime is prepared by slaking which essentially involves the addition of water into a slurry form. Slaking has the additional advantage of rendering the lime more manageable by eliminating dust problems. Calcium carbonate $(CaCO_3)$ is available in both powder and stone forms. Limestone is not as reactive and requires longer residence times than the other forms. However, the application of limestone in anoxic beds for passive neutralisation of acid mine drainage is increasing.

Although lime is the cheapest of the neutralisation chemicals, this advantage is often negated by poor management underground. There is a longer reaction time which is often overlooked in the design of settler feed launders, for instance. In this case, the aim of neutralisation is to raise the feed pH to a level at which the chosen flocculant operates optimally. The pH is often measured too close to the dosing point, with the result that the lime has either not mixed completely or, has not reacted completely. The measured pH is then observed as being too low and additional, unnecessary lime is dosed upstream. Factors such as insufficient mixing due to lime's low solubility, poor baffle and launder design compound this problem. Many mines have installed automatic lime dosing control systems with feed back from continuously monitoring pH probes. The problems of insufficient reaction time and mixing are however, still applicable to automatic control systems. Such a system is further disadvantaged however, by the hostile conditions encountered underground. Humidity and lack of maintenance of the pH probe are the major problems which have too often resulted in failure of the control system and subsequent abandonment.

A further problem with neutralisation as pretreatment for flocculant dosing is the massive variations encountered in feed flowrate to the settlers. Although proportional weirs as described by various authors(18,19) alleviate this problem to some degree, manual dosing, as a response to some downstream pH measurement, is inadequate. The ideal control set-up would include a feed forward loop anticipating variations in flow and pH and adjustment of lime dosage accordingly.

Dewatering operations at ERPM's South West Vertical shaft make use of slaked lime for two purposes. The lime is slaked on surface and a portion is delivered underground to condition the corrosive water before it is sent through the pumps to surface. Once on surface, the lime slurry is used as a pH adjustment for the high density sludge (HDS) thickening process. In this case pH is adjusted to a level where the iron content is precipitated out. The process has wide application and has proved to be effective and economical in acid water treatment(20, 21).

The use of limestone as a cheaper alternative to lime in the neutralisation of acid waters in both the chemical(22) and the mining(23, 24) industries is described.

Table 3 : Characteristics of common neutralisation chemicals

Parameter	Lime	Soda Ash	Caustic Soda
Chemical formula	$Ca(OH)_2$	Na_2CO_3	NaOH
Solubility at 30 °C (g/ℓ)	1.53	568	1 190
Consumption of pure product per g of CO_2	0,85	2,41	0,91
Consumption of pure product per g of H_2SO_4	0.76	2,16	0,82
Increase in hardness per g of CO_2 in g $CaCO_3$/ℓ	1,1	0	0
Increase in hardness per g of H_2SO_4 in g $CaCO_3$/ℓ	1,0	0	0
Product from reaction with CO_2	$Ca(HCO_3)_2$	$2NaHCO_3$	$NaHCO_3$
Product from reaction with H_2SO_4	$CaSO_4$	$Na_2SO_4 + 2NaHCO_3$	Na_2SO_4
Residual buffer capacity when neutralising H_2SO_4	no	yes	no
Reaction time	slow	rapid	rapid
Cost	low	high	high

4.1.2 Soda Ash

Because the reaction time of sodium carbonate or soda ash (Na_2CO_3) is more rapid than that of lime, and because of the high solubility many of the dosing and control problems associated with lime are not apparent. However, consumption per gram of aggressive CO_2 and H_2SO_4 is nearly three times that of lime and the actual cost of the reagent is high. The combination of these two factors make soda ash a very expensive method of neutralisation. A distinct advantage of soda ash is that it imparts a residual buffering capacity to the flow. The inability of lime to add buffering capacity is often overlooked and the low cost is the only basis for reagent choice, which can prove to be *simplistic and short-sighted. Where water is extensively reused and* recirculated, the advantage of the buffer capacity, imparted by soda ash, may compensate for the added cost of soda ash.

4.1.3 Caustic Soda

As with soda ash, reaction time of caustic soda (NaOH) is quick and solubility is high, both factors which facilitate dosing control and mixing. Consumption of pure product per gram of aggressive CO_2 and H_2SO_4 is similar to that of lime but the cost of the raw reagent is high enough to limit the application. An additional disadvantage is the inability to add buffering capacity. Caustic soda is also a hazardous chemical which has logistic and cost implications.

4.1.4 Other Reagents

In cases where the neutralisation is aimed at lowering the pH, namely correction of scaling type waters, treatment of alkaline water before discharge and pH reduction after lime softening, the most common reagents are sulphuric and hydrochloric acid. Usually, however this type of neutralisation is carried out with CO_2, which is a physical process and is discussed in the following section.

4.2 PHYSICAL METHODS

Physical methods use both gas and liquid phase mass transfer operations to alter the carbonate balance and hence change the pH.

The most common use is the removal of CO_2 by stripping from the liquid phase. In the packed tower type of degasifier, the water is sprayed or finely dispersed over a packed bed. A stream of CO_2 - depleted air is fed counter current to the water flow and this serves to strip the gas from the water. A variation is the diffuser degasifier, where the stripping gas is fed into the water through submerged diffusers and bubbled up through the aqueous environment. Vacuum dearators operate close to the saturated vapour pressure of water. This facilitates the transfer of dissolved gas into the gaseous phase. Vacuum dearation is expensive and is usually used in series with a conventional decarbonator stripping column for water with a high CO_2 concentration.

The reverse process of adding CO_2 to water, in order to reduce pH, is also practised but is not as common. The CO_2 is added by injection into the aqueous phase. In order for the CO_2 to stay in solution, the water must be maintained at the required pressure, which is costly.

4.3 COMBINATION METHODS

These methods make use of physical unit operations to ensure the necessary contact required for chemical reaction, and neutralisation, to occur.

4.3.1 Anoxic Limestone Drains (ALD)

Calcium and magnesium carbonate based materials are naturally and commercially available in a variety of grain sizes for use in filter bed type processes. The anoxic limestone drain (ALD) which is being used increasingly in passive treatment systems is such an example. The idea of the ALD was first described and investigated as a viable alternative for acid mine drainage treatment in the early 1990's(25, 26, 27) , although previous studies had indicated possibilities of limestone use(28). These limestone beds are kept anoxic in order to prevent the oxidation of ferrous iron. This in turn limits the armouring of the reactive surface of the limestone with ferric hydroxide. The anoxic technology is relatively new and the chemistry involved in the process is the subject of several detailed studies(29, 30, 31). Guidelines as to the design and operation of ALDs are, however, available(29, 32). The ALD is described more fully in a separate literature review on passive treatment systems.

4.3.2 Reciprocal Neutralisation

This method involves the blending of two streams which have opposing characteristics such as acid and alkaline or aggressive and scale forming flows. Obviously, this type of treatment eliminates the need for costly chemicals and equipment but it is unusual to find two streams that can adequately balance each other and which can be blended without expensive redistribution.

4.3.3 Fluidised Bed with Limestone

An alternative method of contacting the acid water with the neutralising reagent is in a fluidised bed. This method has been tested with various particle sizes of limestone on both a batch and semi-continuous basis. The study investigated the effect of contact time on neutralisation as well as the effect of various cations on the required contact time.

The fluidised bed process allows for the complete utilisation of the limestone but it was shown that the neutralisation rate was dependant on the type of ion in solution. In the case of iron, the neutralisation rate for solutions containing Fe(II) was slower than that for Fe(III). Pretreatment aimed at oxidation of the Fe(II) content, or longer retention times would be required for efficient, full scale operation. An added advantage of the process is the removal of sulphate to the point where the water is saturated with calcium sulphate.

5. CONCLUSION

Neutralisation is used as a water treatment process for the correction of pH generally, as well as adjustment of the carbonate balance which is aimed at stabilising scaling or corrosive waters. Simple pH adjustment may be required for release of the water to the environment or as a pretreatment for a downstream process.

Correct and well controlled neutralisation affords opportunities for significant cost savings through longer equipment life, reduced maintenance or reductions in reagent consumption. However, the chemistry involved in neutralisation is complex and a misinterpretation of this chemistry often results in ineffective process control. This in turn leads to excessive neutralisation chemical costs and the potential benefits are transformed into unnecessary losses.

The application of neutralisation in mine water treatment is mainly in the field of acid mine drainage amelioration and conditioning prior to flocculant dosage in the settling and thickening processes. Chemical dosage, in the form of slaked lime, is the most common method of neutralisation, due to its cost effectiveness. For acid mine drainage treatment, there is increasing application in the field of passive or low maintenance treatment with the use of anoxic limestone drains.

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COAGULATION, FLOCCULATION AND SETTLING

1. INTRODUCTION

One of the most common problems in process and service water is the suspended solids content. The separation of the solid and liquid contents is required for several reasons, depending on the application. In South Africa as in most arid countries, there is a high degree of water reuse and recycle. Solids will accumulate during each cycle until the resultant slurry becomes unfit for use. In addition the abrasive nature of solid particles imparts additional wear on piping, pumps and process equipment. The removal of the solids becomes essential if the quality and usefulness of the water is to be maintained. In other applications, it is the solids content which is valuable and the further treatment thereof requires it to be in a relatively dry form.

The mining industry encompasses various aspects of separation but most involve the recovery of a clean water product. Underground operations require large volumes of water for dust suppression, cooling, hydro-power and in some cases water-jetting. There is a high degree of solids accumulation in these processes and the water reports to underground sumps for delivery to surface with a high suspended solids content. In order to protect the pumps and water columns, it is necessary to remove a large portion of the solids content. Once on surface, the metal recovery process requires the solid material to be crushed and pulped with water for downstream leaching. After metal removal, the disposal of the barren slurry is facilitated by thickening and removal of excess water. The slurry or slime is then pumped to a slimes dam where solids settling takes place and there is additional water recovery.

Although it is often measured and interpreted as a single contaminant, this is an oversimplification and the suspended solids usually comprise a variety of individual species and particle types and sizes. A knowledge of this makeup is essential in the development of treatment systems aimed at the efficient separation of the solid and liquid fractions. Colloidal particles, apart from being small enough to pass through conventional filtration media, have the added disadvantage of stability.This prevents the collision and agglomeration which would render the resultant larger particles, settleable by gravity. Efficient coagulation and flocculation requires a knowledge of the type of colloidal particles so that the stabilising properties may be suppressed or even overcome. The principles involved in coagulation and flocculation are discussed in a separate section.

Once a degree of particle aggregation has been achieved, it becomes possible to separate liquid and solid contents under the influence of gravity or other applied forces. This is achieved in a variety of processes including thickeners, clarifiers, settlers or settling dams. The correct design of such facilities must allow for the protection of the particle flocs and must create suitable conditions for these flocs to settle.

2. COAGULATION AND FLOCCULATION

It is useful at the outset to define the terms coagulation and flocculation as they are often used interchangeably, which is not strictly accurate. The overall process of floc formation aims at the creation of suitable conditions for particle aggregation. Initially, the stabilising repulsive charges on the particles must be neutralised. The particles must then be brought into contact with each other so that flocs may form and settling is enhanced. It is common to refer to the first step as coagulation and this may include the processes of double layer compression and charge neutralisation. Flocculation describes specifically the aggregation of particles through mechanisms such as bridging and enmeshment. These processes are described more fully below.

2.1 OBJECTIVES

The process of coagulation and flocculation is intended to condition the water and the suspended solids content therein, such that the formation of larger particles or flocs is encouraged. These larger, and consequently heavier, particles are then more easily settled by downstream processes such as thickening, clarification and settling. The need for coagulation and flocculation is clear when a comparison of various particle settling times is made. Some of these are presented in Table 1. The settling times are based on Stokes' Law.

This table indicates that it is not practical to settle particles smaller than 0.01 mm in diameter. This is particularly so in situations where the area available for settling is limited. This limitation occurs in the confined spaces of underground mines. The aim of coagulation is thus to reduce the required settling time of particles and thereby reduce the surface area required for a particular solid loading rate. The principles of the settling process itself are described in Section 3.2 below.

Table 1: Settling Times Of Various Particles And Particle Sizes

Particle diameter, mm	Particle type	Time to settle through 1m depth	Particle specific area, m^2/m^3
10	Gravel	1 second	$6x10^2$
1	Sand	10 seconds	$6x10^3$
10^{-1}	Fine sand	2 minutes	$6x10^4$
10^{-2}	Clay	2 hours	$6x10^5$
10^{-3}	Bacteria	8 days	$6x10^6$
10^{-4}	Colloid	2 years	$6x10^7$
10^{-5}	Colloid	20 years	$6x10^8$
10^{-6}	Colloid	200 years	$6x10^9$

2.2 MECHANISMS OF COAGULATION AND FLOCCULATION

Coagulation, as defined above, is aimed at the destabilisation of solid particles. This destabilisation involves the elimination or reduction in inter-particle repulsive forces. Once the particles are destabilised, natural or induced motion will move particles together and the aggregation can occur.

2.2.1 Coagulation

There are two main mechanisms of particle destabilisation. Both involve the reduction of the surface potential which is responsible for the repulsive forces.

2.2.1.1 Double-Layer compression

Particles possessing a fixed surface electrical charge also require a counter charge to balance and neutralise the surface potential. This counter charge is comprised of a number of ions which are concentrated in a diffuse layer, called the double layer, surrounding the particle. When an electrolyte, in the form of a coagulant, is added to the colloidal dispersion, the density of the charge in the double layer is increased. This has the effect of reducing the necessary volume for charge neutralisation. The implication is that the depth of the double layer is reduced or compressed. The van der Waals attractive forces between particles then become more dominant and aggregation is enhanced. The net attractive force is a sum of these van der Waals forces and the electrical double layer repulsive forces. At high concentrations of electrolyte, the double layer is reduced dramatically and the van der Waals forces are correspondingly extremely dominant. Aggregation thus occurs very rapidly. The practical implication of this observation is that increasing the electrolyte concentration causes an increase in the rate of aggregation.

2.2.1.2 Charge neutralisation by adsorption

Certain chemical species may be adsorbed directly onto the surfaces of colloidal particles. If the adsorbed species has an opposite charge to that of the particle, then charge neutralisation will occur.

The knowledge of which mechanism is responsible for coagulation is important as it has implications for the concentration and dosing of the coagulant. It is clear that the mechanism of adsorption destabilisation is stoichiometric which implies that as the concentration of colloids, or colloid surface area increase, so the required dosage of electrolyte increases. Overdosing in the case of double-layer compression causes an increase in the rate of particle aggregation, while overdosing of adsorbable species can actually retard the aggregation process. This is due to the restabilisation of particles when the surface charge is reversed to that of the adsorbed species.

2.2.2 Flocculation

Aggregation of the destabilised colloids can only occur if there is contact between the particles. This can occur naturally or may be induced mechanically. The thermal or Brownian motion present in a liquid gives rise to perikinetic flocculation. Particle contact induced by stirring or differential settling is termed orthokinetic flocculation.

An analysis of the flocculation rates obtainable via the two mechanisms shows that perikinetic flocculation is dominant at particle sizes $< 1\ \mu m$. It is also clear, however, from Table 1 that particles of this diameter do not settle very quickly. These two factors imply that some form of orthokinetic flocculation is required for effective settling. Usually this is achieved by baffles, mechanical mixing or agitation with compressed air.

The degree of mixing or agitation in a flocculation basin can be quantified by measurement of the mean velocity gradient which is a function of the basin volume, fluid viscosity and power input to the basin. The velocity gradients need to be high enough to allow sufficient particle contact but must not be so high as to cause the shearing of the newly formed flocs. This is particularly problematic in the case of organic polymer flocculants because once the long polymer chains become detached, they may fold back around the particle and restabilise it.

Although the flocs formed by iron(III) and aluminium(III) salts are not as strong as the polymer flocs, it has been found that velocity gradients of between 25/sec and 100/sec are suitable for the metal salt flocs while lower values such as 15/sec are optimum for the polymer floc formation.

Another important criteria in the design of flocculation systems is the detention time as this is a measure of the opportunity for contact to occur.

2.2.3 Other mechanisms

Another mechanism which combines coagulation and flocculation and which does not involve charge neutralisation is the sweep-floc process in which colloids become enmeshed in metal salt precipitates. In some cases the colloids form the nuclei for the precipitation of the insoluble metal hydroxides.

Some large molecule natural compounds such as starch and cellulose are also known as coagulating agents. Various theories have been developed to explain the mechanism and it is accepted that the long chain molecules act as a bridge between two or more colloidal particles to form the beginning of a floc.

3. SETTLING

The process of settling is used in various applications for the separation of solid and liquid contents. The degree of separation may range from the recovery of water from process slurries to the clarification of colloidal waters for potable use. The main mining applications include the thickening of plant residue, prior to the disposal on slimes dams and the removal of suspended solids from underground service water before delivery to surface. Underground settling forms the focus of this discussion since it incorporates many of the general principles involved as well as presenting special difficulties, as discussed below.

3.1 OBJECTIVES

The general objective of any settling process is the separation of suspended solid material from the aqueous media. Although it is essentially a physical process relying on the force of gravity, there are chemical considerations involved in the coagulation step as well as in the control of the overall process. This is discussed further in Section 3.4.

The specific objectives of settling underground service water are two-fold:

- Removal of solid material from the water prior to pumping, in order to reduce the abrasion of pipes and pumps.
- Separation and collection of sludge content for delivery to surface and treatment in the metallurgical plant. Typically the sludge has a high grade of gold.

It is difficult to quantify the cost of increased wear on process equipment due to the presence of suspended solids, but it has been estimated that costs which can be ascribed to poor settling alone are approximately R140 million. Combined with the increase in gold recovery from the sludge, the importance of efficient underground settling becomes clear.

3.2 THEORY AND DESIGN OF SETTLERS

The velocity, V, at which a particle, of diameter d, will settle in a fluid is given by Stoke's Law as follows:

$$V = \frac{2gd^2(\rho_1 - \rho_2)}{9\eta} \qquad (1)$$

where ρ_1 = density of the particle
ρ_2 = density of the fluid
η = coefficient of fluid viscosity

Application of this law for various particle sizes yields the settling times presented in Table 1.

Apart from the particle settling velocities, the other critical parameters involved in the design and operation of any settling installation are:

- hydraulic or surface loading rate ($m^3/m^2.h$)
- solids loading rate ($kg/m^2.h$)

Due to the cyclical nature of underground mining operations these criteria fluctuate widely and often rapidly. The peak in the hydraulic loading can be up to 10 times the baseline flow. In addition, the maximum hydraulic loading corresponds to an increase in suspended solids content of between 100 and 400 times the normal concentration. This combination leads to peaks in the solids loading rate of up to 4000 times the baseline value. It is clear that with fluctuations such as these, good efficient control of settling is essential. This aspect is discussed further in Section 3.4.

The flowrate to any underground settler is usually a factor of the total tonnage of rock mined per unit time, T. *The volume of water required per mass of rock* mined, W_T , differs for various operations but an industry average figure of approximately 2 m^3 per ton is applicable. The maximum design flowrate, Q_m, should allow for double this calculated flow and this gives the expression:

$$Q_m = 2(W_T)(T) \qquad (2)$$

Calculation of the settling velocities from equation (1) will give a minimum required settler surface area, A_s , as follows:

$$A_s > Q_m / V \qquad (3)$$

This surface area excludes that required for feed and overflow launders.

3.3 TYPES OF SETTLERS

The variables of design in settler systems are the hydraulic and solids loading rates and these are dependant on the choice of settler type and configuration. A summary of the typical hydraulic or surface loading rates for various configurations is presented in Table 2. The various types of settler *configuration are then discussed.*

Table 2: Surface Loading Rates For Various Types Of Settlers

Settler Type	**Loading rate, $m^3/m^2.h$**
Conventional shallow rectangular	1 - 4
Conventional deep conical	5 - 8
Conventional sludge blanket	5 - 10
High rate sludge blanket	15 - 20
Inclined plate or lamella	15 - 20
High rate tube	50 -60

3.3.1 Horizontal flow

The simplest form of settlers consist of shallow cells in which the dirty water moves very slowly in a horizontal direction from inlet to outlet. The basis of design is to allow sufficient detention time for particles to settle before the outlet is reached. The bottom is slightly inclined to allow for the collection and removal of sludge. This type of settling installation is usually found at the older shafts. These installations may operate in a batch manner in stagnant sumps or with continuous introduction of influent and removal of sludge.

3.3.2 Lamella or inclined plate

Lamella plate type separators consist of a number of parallel closely space inclined plates over which the dirty water is fed. Gravitational separation occurs and the clarified water collects under each plate and continues to flow towards the top. Settled sludge collects along the bottom of each plate and flows down the incline to be collected in the underflow. The lamella separators have wide application in wastewater treatment but are not particularly suited to the wide fluctuations in solids and hydraulic loading encountered in underground settling operations. There is also a tendency for plates to block.

3.3.3 Vertical flow

The more modern settlers operate on a vertical flow basis with the introduction of the influent dirty water occurring at some depth below the settler surface level. Originally the idea was developed for use in the sewage treatment industry and later developed for mine water treatment. The various types of configuration include double-V, cylindro-conical, high rate tube and sludge blanket settlers. Conventional vertical flow settlers operate on the basis that the rising velocity of the water is lower than the settling velocity of the flocs, giving a net settling out of the solids. Vertical flow settlers require substantially less surface area than horizonatal flow settlers. The principle of operation for the double-V, the high rate tube and sludge blanket settlers makes use of orthokinetic flocculation as described previously. Influent water is introduced at some depth below the settler surface which must be below the floc or sludge blanket which forms in the settler body. This water then rises to the surface and the associated suspended solids encounter the fluidised sludge blanket. Orthokinetic flocculation occurs and a portion of the flocs increase in size, thus increasing their settling velocity and they fall out of the sludge blanket, to be collected at the bottom of the settler cell.

The important aspects of such settler arrangements are the uniformity of the influent feed and clear water withdrawal as well as sludge removal. Ideally, there should be no local velocity gradients or short circuiting which tend to disturb the sludge blanket. This is achieved by introducing the dirty water and withdrawing the clear effluent across the entire surface area of the settler. A uniform upward flow over the settler area is thus created.

3.3.4 Ideal characteristics of underground settlers

The progress in development of underground settlers has been necessitated by the unique conditions and restrictions encountered underground. The ideal underground settler should have the following characteristics:

- *High rate which reduces the required surface area*
- Ability to cope with wide and rapid fluctuations in feed flow and solids loading
- Low maintenance and operator skill requirements
- Robust and correct control of lime and flocculant dosing.

These aspects are discussed briefly in Section 3.4

3.4 OPERATION OF SETTLING SYSTEMS

As with any process the operation and control of settling systems is facilitated by a knowledge of the principles and mechanisms involved. The settler cell cannot be controlled in isolation and aspects such as feed launder flow, pH adjustment, flocculant dosing, mixing and influent introduction are integral components of any efficient settling operation.

In cases where the settler system has been overdesigned, the control and distribution of flow during peak periods is easily achieved with the use of gate valves in the feed launders. Ideally, each settler should be maintained under constant conditions and fluctuations in feed flow should be smoothed out where possible.

The adjustment of pH by the addition of neutralising chemicals is often required for conditioning prior to flocculant addition. The control of this neutralisation step is often poor. The target pH is thus seldom achieved and this results in the poor performance of the dosed flocculant. The problems of underground neutralisation are discussed more fully in that particular literature review.

The problems related to flocculant dosing are similar to that of neutralisation in that there is seldom a correlation between flocculant dosage and dirty water flowrate or suspended solids concentration. The design of the feed launders from the dosing point is critical since excessive turbulence causes flocs to shear. The launder should be designed so as to allow a period of initial mixing and gradual turbulence reduction until the point of introduction into the settler cell.

4. CONCLUSION

The importance of water and solid separation in industry and wastewater treatment is well understood. Settling under the influence of gravity provides a relatively inexpensive method of solids removal from large flowrates of turbid water and slurries.

Flocculants and coagulants are added to waters containing particles with very low settling velocities. The knowledge of coagulation and flocculation mechanisms is important if the optimum conditions are to be created.

Underground settling of dirty service water is carried out in order to reduce abrasive wear on pipes and pumps as well as to recover the high gold grade solids content which is washed down from the working areas of the shafts. Underground conditions are particularly hostile and the high cost of development limit the space available for settling. This fact has prohibited the use of horizontal flow settlers and recent trends are towards vertical flow configurations such as cylindro-conical and high rate tube settlers.

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FILTRATION

1. INTRODUCTION

The mineral extraction industries utilise vast quantities of water during the processes of mineral beneficiation. Restricted water supplies have precipitated the need for water conservation, and naturally focuses on the optimisation of process water recycling.

The best techniques for the removal of suspended solids from water depends on a number of factors including particle size, solids concentration and flowrate. Correlation's exist which broadly define regimes in which different physical purification methods are most suitable(1,2,3). Mine waters are characterised by high levels of suspended solids as well as high dissolved solids content. The bulk of suspended matter can be removed from mine water efficiently and cost effectively through the current practice of sedimentation. However, specialised mining applications, cooling systems and high pressure pumps for hydraulically powered mining machinery require higher water qualities to prevent particle erosion and scale formation. The particle size of solids in the water emerging from settlers is generally less than 30 µm, and these particles have to be removed by filtration.

Mine waters are often contaminated with chemicals for various reasons. Continuous recycling of the water and evaporation, coupled with geochemical leaching from the ore results in an increase in concentration of sulphates, chlorides and carbonates. The addition of lime for pH control and the use of corrosion and scaling inhibitors contribute to the flocculation of the water. These bridge structures linking the suspended particles often lead to premature fouling of filters, and hence high load filters are necessary(3). Filtration systems thus have to be selected to efficiently cope with the demands of influent water clarification to meet effluent water specifications.

2. PRETREATMENT TECHNIQUES FOR SLURRY FILTRATION

The physical properties of any sludge or slurry are governed by the nature of the particles in the dispersion phase, and the size distribution of the particles. In general, sludges or slurries containing predominantly large particles form an open structure allowing high liquid flowrates and consequently rapid rates of thickening and filtration. However due to the open structure, such suspensions often have poor packing characteristics and therefore low solids concentrations either in the thickening sludge or dewatering cake. Conversely, suspensions containing many small particles will have good packing characteristics, but low rates of thickening and filtration.

Pretreatment of the particles modifies the size distribution either to improve the packing characteristics or to increase the voidage in a suspension to allow increased rates of thickening and filtration. The attachment of a suspended particle at the solid/liquid interface presented by the filter can be controlled by the surface properties of these solids(4). Both the particle and the filter grain react with the inorganic and organic dissolved species in solution. Both solids have an electrical charge which is balanced by accumulations of solute ions of opposite charge arranged in compact and diffuse layers near the solid surfaces so that each interfacial region is electrically neutral(14). In practice this would allow greater throughputs using any given filtration plant. To optimise the thickening or filtration system, the increased operating costs incurred by the pretreatments process must be offset against any reduction in capital costs resulting from greater throughputs(17).

2.1 CHEMICAL PRETREATMENT

2.1.1 Metal Salts

Metal salts are the simplest form of chemical pretreatment substances, of which aluminium and iron salts are the most common. Aluminium or iron salts are added in sufficient dosages to form voluminous amorphous precipitates of $Al(OH)_3$ or $Fe(OH)_3$. Solid material is formed in the water to provide targets or contact sites for the flocculation of the natural particles in suspension.

Aluminium sulphate is used for many types of suspensions, while the chloride form is more effective for organic sludges per mass of aluminium added. Aluminium chlorohydrate may be used as an alternative to aluminium chloride, as it has the effectiveness of the chloride, while minimising the substantial lowering of the pH caused by the hydrolysis of the chloride to hydrochloric acid at high dosages.

The chloride forms of iron are also more effective than the sulphate forms, while the trivalent ferric salts are more effective than the divalent ferrous salts. Ferrous sulphate is relatively inexpensive and so is commonly used in sewage treatment. Lime is often used with iron salts to raise pH, ensuring that iron is precipitated as the hydroxide. The pH is not necessarily the controlling factor, although the charge on the particles may be influenced by the pH.(17) The lime appears to have a specific role in the formation of a matrix around the particles, in addition to raising the pH. When using lime and ferrous sulphate of ferric chloride, it is normal practice to add the iron salt first, followed by the lime addition.

2.1.2 Polyelectrolytes

Organic polyelectrolytes are widely used alternatives to metal salts, particularly for packed bed filters, belt press filters, centrifuges and plate filter presses and vacuum filters. An advantage of polyelectrolytes over inorganic coagulants is that they do not significantly add to the mass of the cake.

Polyelectrolytes are added to the water to destabilise the particles and little or no additional solid volume is formed. Polyelectrolytes are long chain organic molecules with molecular weights of up to 20 million, having at intervals along the chain, ionic groups. The polymers from which the polyelectrolytes are made are usually polyacrylamides or polyamines, which carry both positive (cationic) and negative (anionic) charges. The polymer solutions are usually dilute (0,05%-0,5% of active product by mass), since in concentrated solutions, the charged sites on the polymers tend to cause the molecules to curl up, hindering the efficient adsorption onto the particles. When the polymer solution is dilute, the polymer molecules will uncurl, allowing more of the charged sites on the molecule to be used. The mechanism of polyelectrolyte flocculation is discussed in detail in numerous texts[6,7,8].

2.2 PHYSICAL PRETREATMENT

Physical pretreatment techniques are sometimes used instead of, or in addition to chemical pretreatment. Physical pretreatment techniques include freezing and thawing, elutriation and irradiation, which can be used to improve the filterability of sludges. Physical pretreatment techniques are either expensive (freezing and thawing) or only partial treatment methods (elutriation), and are consequently only suitable for specialised situations (eg. where chemical additions are not possible due to the importance of the filter cake purity)[17]. A useful summary of pretreatment techniques is given by Hermia[10].

3. PRETREATMENT TECHNIQUES FOR WATER FILTRATION

The primary pretreatment applied to water before filtration is flocculation and sedimentation. This pretreatment will typically reduce the suspended solids in the water to be filtered to 10-20 mg/ℓ. More details on this pretreatment are given in a separate literature review.

4. FACTORS AFFECTING FILTRATION

Filtration refers to a broad spectrum of processes to which a generic grouping of common factors affecting performance is relatively difficult to compile. In general the size, distribution and concentrations of suspended particles, the nominal velocities of the fluid phase passing through the filter, surface holding capacity of solids and solids loading capacities will control the performance of filtration. The principles of these statements are illustrated by considering certain water treatment filtration processes.

4.1 SCREENS

These remove larger types of solids such as debris, rags, grass, fish and so on. Screens are normally installed to protect downstream equipment.

4.2 STRAINERS

These in-line installations remove finite particles and foreign material which may pass through the screens. The nominal aperture of the wire mesh determines the performance of the strainer.

4.3 MEDIA FILTERS

These remove the smaller sized range of particles by passing the feed water through a bed of filtering media (e.g. sand filters). Media filtration is the most common form of filtration.

4.4 FIXED MEDIA FILTERS

These polish water to an absolute filter pore size rating (e.g. cartridge filters, cross-flow micro filters, ultra filtration units and membrane filters).

Potential solids holding capacity can be estimated as follows:

$S_h = 0{,}11 \times D_s$

Maximum permissible suspended solids concentration in influent can be calculated as follows:

$S_{mp} = (S_h \times F_b \times 1\ 000) / (F_r \times F_c)$

D_s = max. solids concentration in influent water after 24 hours settling (kg/m^3)
F_b = filter bed depth (m)
F_c = desired filtration/backwash cycle (h)
F_r = filtration rate (m^3/m^2.h or m/h)
S_h = solids holding capacity of filter bed (kg/m^3)
S_{mp} = maximum permissible suspended solids in influent (mg/ℓ)

5. TECHNOLOGIES FOR WATER FILTRATION

5.1 DEEP BED FILTRATION

Deep bed media filters are used for the clarification of liquids to a very high degree of purity. Normally suspensions to be clarified are of relatively low concentration, usually less than 50 mg/ℓ, and never greater than 500 mg/ℓ(18). Media filtration is generally a discontinuous process. As particles accumulate on the filter medium, the energy input to move the fluid through the interstitial spaces increases until a limiting energy dissipation rate is reached, and a filter cleaning or replacement is initiated. Filtration mechanisms in deep bed filters include straining and non-straining processes. Straining occurs when the particles are larger than the pore openings in the filter bed. The efficiency of the process is not determined by surface chemistry, and the use of coagulants is not necessary to achieve suspended particle removal by this mechanism. Removal of particles smaller than the pore size of this medium requires transport of the particles from the fluid to the surface of the medium or a previously deposited particle. As the particle approached the surface, short range surface forces influence the particle transport. Attachment of the particle to the surface occurs if the short range repulsive forces are minimised (i.e. the interacting surfaces are destabilised). Three transport mechanisms occur during filtration, interception, sedimentation and Brownian motion.

The filtration process is essentially a first order process, and therefore increasing the influent concentration has no effect on the suspended solids removal efficiency, this does however require increased backwashing frequency until a point is reached where the filter's water recovery becomes unacceptably low. Increasing the suspended particle density enhances the transport by sedimentation, and thus increases the removal efficiency.

Table 1 : Important Variables in Granular Media Filters

Variable	Range	Comments
Filter medium		
• media type	• single medium (sand or anthracite) • dual media (sand and anthracite) • multi media (sand, anthracite and garnet)	Dual or multi media have better bed penetration, higher solids holding and longer filter runs
• media grain size	• sand: 0.35 - 2 mm • anthracite: 0,8 - 4 mm	Decreased grain size generally gives better solids removal but higher head loss
Filter bed		
• solids holding capacity[1]	• increases when moving from gravity to pressure filters and single to dual or multi media • granular media filters contain 45 % pore volume with 11 % of bed volume available to hold solids	Solids holding capacity of around 20 kg/m^3 should be possible for mine water filtration with up flow or pressure filter
Filter operation		
• filtration rate	• typical 5 - 25 m^3/m^2.h	Affects size of required plant. Serious deterioration in product quality if safe maximum is exceeded
Filter cleaning		
• backwash cycle	- combined air and water backwash • air flow : 50 - 60 m^3/m^2.h • water flow during air scour : 5 - 10 m^3/m^2.h • water flow during rinse : 15 - 25 m^3/m^2.h • water recovery : 98 - 99 % - air followed by water backwash • air flow : 50 - 80 m^3/m^2.h • water flow : 20 - 50 m^3/m^2.h • water recovery : 95 - 97 %	Combined air and water backwash is preferred to water only. Air followed by water backwash is used for dual/multi media filters. Maximum wash rates will depend on media grain size and density
Influent quality		
• suspended solids conc.[2]	• rule of thumb : < 50 mg/ℓ	Affects product quality, backwash frequency, water recovery

Increasing the length of the filter bed essentially increases the length of the filter pores, and the efficiency of particle attachment increases, thus improving removal efficiency. The removal efficiency decreases with increasing interstitial velocity and the diameter of the filter pore which is proportional to the grain size in granular media beds.

5.1.1 Slow Sand Filter

The slow sand filter is a special type of sand (media) filter characterised by low capital cost and simplicity of operation(36). The filter is operated at low flowrates (less than 0,2 m^3/m^2.h), and the low flowrate yields efficient removal of large particles (those affected by interception, sedimentation and straining mechanisms). The filter bed usually consists of a fine, unstratified sand medium, resulting in the deposition of most particles within the top section and on the surface of the filter bed. Since the rate of filtration is low, the resultant head loss is usually unacceptable. Slow sand filters are operated without filtrate treatment by coagulant addition, which is one of the advantages of slow sand filtration, however very few sub-micron particles are removed. The sub-micron particles diffuse towards the stationary surfaces, but without destabilisation, attachment does not occur. This type of filter is relatively ineffective for the clarification of high turbidity waters. The media is cleaned by backwashing clear filtrate back through the bed at high velocities to scour the deposited particles from the surfaces of the granular media.

5.1.2 Rapid Gravity Filter

These filters utilise the principles of head loss reduction and deep bed retention of suspended solids to increase filtration rates. The rate of development of head losses in granular filter beds where suspended particle deposition occurs within the interstitial voids depends on the rate of particle volume removal, the filtration rate, and the size and total volume of the interstitial spaces. The lowest rates of head loss and the longest filter runs occur when the region in which deposition takes place is deep (long pore lengths) and the medium grain size is large (large pore diameters).(36, 42, 48) The efficiency of the filter can be manipulated by media grain size selection, media composition (sands and anthracite)(1, 3, 37, 38, 48), media stratification and flow orientation (upwards or downwards). These filters in various configurations: single media, dual media, multi-media and up flow are widely used for the treatment of mine service water(1, 3, 34, 37).

5.2 PULSED BED FILTRATION

The pulsed bed filtration system utilises a shallow bed of fine grained sand to capture non-flocculated suspended solids. A pulsing system is used to periodically regenerate the surface of the bed. The regeneration of the surface of the filter medium bed allows the acceptance of high levels of suspended solids loadings without surface blinding, and thus provides economic filter cycles(26).

5.3 CONTINUOUS SAND FILTRATION

The removal of suspended solids and turbidity is often accomplished with the use of rapid sand filters. After a certain amount of suspended solids have been removed by the filter, the increase in pressure drop and the breakthrough in suspended solids requires a media wash or backwash cycle to completely clean and restore the media. A variety of designs exist which extend the filtration cycle. However, the principle problems in filter operation are associated with maintaining the filter bed in good condition. Backwash systems are designed to provide high velocity backwash efficiency. The backwash cycle is usually complex and expensive due to auxiliary equipment associated with it and requires the filter to be shut down during the cycle. Continuous filters offer several advantages over conventional batch backwashed systems. Continuous filters use an internal airlift and sand wash system which allows the sand to be constantly recirculated, which in turn allows continuous filtration(27, 28, 29). The continuous filter is a granular media, up flow gravity filter. Two operations occur simultaneously in the filtration vessel. While the influent suspension is filtered, the dirtiest sand in the filter is internally cleaned and returned to the filter bed. The influent suspension enters the filter via a series of vertical pipes (riser tubes) which are spaced to allow even flow into the filter. These vertical pipes discharge under a distribution hood, which uniformly introduces the influent into the sand bed. The distribution hood is positioned so that a stratum of media lies above the distribution point. An area below the inlet exists to store and transport the dirty sand used in filtration. The distribution hood is designed to allow the media to move downward, past the influent distribution point without interference, The suspension is filtered upward through the sand or media. Suspended solids are removed via depth filtering principles and the clean filtrate exits over a filtrate weir located at the top of the filter.

An airlift is centrally located at the base of the filter, which constantly discharges small amounts of air to continuously convey a slurry of sand and filtered solids from the bottom to the top of the filter tank. This allows the entire sand bed to constantly move downward, so that the dirtiest sand is conveyed upward by the airlift for sand washing. The action within the airlift is very turbulent and the filtered solids are scoured off the filter media. Backwash rates are high (up to 10 times conventional backwash rate) and are compounded by air backwashing, resulting in extremely efficient media cleaning(27).

Due to the counter current mode of operation, and the continuous regeneration of the filter media, the formation of a layer of filtered solids in the direction of filtration is avoided. The pressure drop thus remains very close to that of a clean filter bed. Due to the sand bed being in a relatively clean state during operation, the media has the capacity to operate with high levels of suspended solids and operate continuously(27).

5.4 LOW HEAD FILTERS

The low head gravity filter operates according to similar principles encountered in other rapid gravity filters, but the construction differs significantly. The bed is subdivided into strips by partitions at regular intervals, typically 200 mm or 300 mm across the width of the filter, making up a number of cells (up to 160). These cell divisions extend above the media, but always remain below the water surface, thus all the cells are hydraulically connected. The construction of the cells is such that each has its own outlet port through a partition wall which separates the filtered water channel from the main filter body. A travelling bridge wash water system is mounted above the partitions which moves slowly across the length of the filter and is configured to clean each filter element while connecting to the outlet port of each cell. The collecting hood above the filter cell collects the dirty wash water together with a little water from either side of the cell being washed. Whilst the cell cleaning process in is progress, the other cells continue to operate normally, and the loss of throughput of the cell does not affect the total filter performance as it represents less than 1 % of the total filter area(30).

The backwash system efficiency is attributed to a two pump system which pumps filtered water to fluidise the media, and positively removes impurities using the pump attached to the collection hood. The media surface is simultaneously scoured by rakes which ensure that resettlement of wash impurities does not occur. The advantages of the system reflect on the economy of low head requirements for the filter as a whole and the benefit of operating the filter while backwashing is in progress.

5.5 BUOYANT MEDIA FILTRATION

Buoyant media filtration refers to filtration in which the liquid to be filtered is passed up through a bed of granular media (low density polyethylene or polypropylene) that has a specific gravity less than that of the liquid being filtered(19, 20).

The buoyant or floating media filter is essentially a hybrid system incorporating gravity settling and up flow filtration. Buoyant media filtration occurs through a bed of floating media located above the settler section. The construction of the filter includes a stilling well for influent injection into the settling section. A sludge hopper is designed into the base of the filter as a conical section, which assists in sludge removal. The buoyant media is retained by a perforated screen which serves as a physical barrier between the filter contents and the clarified filtrate. In most applications 90% of the solids are removed by gravity settling, while the filter section polishes the overflow to yield a filtrate of very low residual turbidity(19). These filters have been demonstrated to perform exceptionally efficiently, even under high solids loading (up to 2600 mg/ℓ)(19).

5.6 CARTRIDGE FILTERS

Cartridge filtration is usually applied to the removal of low concentrations of suspended particles from low flowrate streams. Cartridge filters are used for the polishing of liquids to a high degree. This implies that the filter assembly should be totally inert to the fluid it contacts, and should not contribute any material from its components to the effluent fluid. Cartridge filters are broadly categorised into depth filters and surface filters. Depth filters are generally manufactured from wound fibre, sintered plastics in packings and loose fill. The primary consideration in filter design is the ability to hold suspended solids without blockage. Surface cartridges usually only have nominal suspended solids holding capability, which is governed by the total surface area of the filter and the depth of solids which can be built up without exceeding the permissible pressure drop(1, 33).

Cartridges are generally replaced when they become clogged, although some types can be cleaned and reused, however the cleaned filters normally operate with reduced capacity due to irreversible blinding of filter pores. The cartridge filter life reduces exponentially with feed solids content, and thus these filters will generally find use in mining as a final stage of filtration, upstream from sophisticated machinery, requiring high quality water (e.g. hydraulic equipment).

5.7 CROSS FLOW MICROFILTRATION (CFMF)

The technique of cross flow microfiltration (CFMF) potentially enables the continuous separation of solids from suspensions, at conditions which approach steady state. The suspension is pumped into a porous tube. The clear liquid permeates the tube wall and is recovered as the permeate, while solids are convicted to the tube wall to form a fouling layer, the cake. The filter cake, in constituting the hydraulic resistance to permeate flow, decreases the permeate flux. However, the bulk flow of the suspension tangential to the cake tends to limit the growth of the cake, eventually resulting in a cake thickness, and a permeate flux which is relatively constant with time. The technique of utilising a tangential flow to minimise growth of a fouling layer is widely used in other membrane separation processes, such as reverse osmosis, ultra filtration(21) and nanofiltration(22). The differences between the processes essentially resolves to the size of the rejected species.

The separation medium in cross flow microfiltration distinguishes two classes of microfiltration, those in which the tube wall is the filtration medium and those in which the cake is the actual separation medium. In systems which utilise the tube wall as the separation medium, support systems include porous stainless steel, ceramic and rigid plastic tube. In these systems, cake formation is an undesirable phenomenon, leading to low fluxes and possibly irreversible fouling of the pores in the tube wall(23).

Systems employing woven fibre tubes, rely on the cake formed by the deposition of solids onto the tube wall as the filtration barrier. The close packing of the cake enables the retention of the particles which may be significantly smaller than the pores in the fabric. The advantage of this configuration pertains to the relatively large fabric pore size which may be used, enabling simplified cleaning procedures and minimising irreversible fouling of the pores(24). Tube diameters range from 12mm to 40mm, permitting operation on suspensions that contain relatively coarse solids, such as mine service water. The tubes may be produced relatively inexpensively in long lengths, which extends the economic viability of CFMF to high volume, low product value operations. CFMF thus has potential for the production of very high quality water from recirculated mine service water(1).

6. TECHNIQUES FOR SLURRY FILTRATION

6.1 ELECTROFILTRATION

Electrofiltration is a technique for filtering ultra fine particle slurries, while achieving significant reduction in energy consumption and capital investment by combining vacuum filtration and electrode position.(25) Electrofiltration is *particularly suited to slurries with a high content of ultra fine or colloidal particles (~70% below 2µm in size). The electrofilter essentially allies vacuum* filtration with electrophoresis and electroosmosis. When a well dispersed slurry of fine particles is placed in a d.c. electric field, the particles migrate towards the anode (electrophoresis) since they have a net negative charge on their surfaces. The negative charge on the particle surface is usually an intrinsic property of clays and organic (coal) particles, or the result of organic dispersants used to deflocculate the solids in the water. The migrating particles deposit on the anode membrane forming a densely packed cake of low water content. The cake is further dewatered through electroosmosis, a process by which excess water is pumped out through the negatively charged capillaries of the cake.

The cathode whose surface is covered with a filter cloth is used as a conventional mechanical filter. A vacuum applied to the filtrate side of the cathode causes a thin cake to form on the surface of the filter cloth. The cake acts as a precoat to trap fine particles and to prevent them from slipping through the pores into the filtrate stream. Furthermore, just as for the anode, the electroosmotic pressures in the cathode cake operate in conjunction with the vacuum pressure, enabling the electrofilter to produce a clear filtrate at higher rates than conventional vacuum filters of comparable filtration area(25).

6.2 BELT FILTRATION

Mechanised methods of coal mining often produce situations where up to 20% of the total mined coal may report as -0.5 mm particles, of which up to 25% may be discarded as fine tailings(31).

The disposal of coal tailings into natural water systems has become increasingly environmentally unacceptable, resulting in widespread use of the filter press as a dewatering system. Belt filtering systems cite many advantages over filter press installations, in terms of capital cost , continuous operation, lower maintenance costs, lower power consumption, however chemical pretreatment of the feed stream is generally necessary(31, 32) .

The belt acts as a support for the matrix of material being processed which itself acts as a filter. Since processing can only be carried out on material having some degree of mechanical stability, due to the shear imposed on the flocs, some form of pretreatment is usually necessary. The sequence is usually pretreatment with polyelectrolytes, thickening and belt pressing. The filter press operates by a process of continuous shearing action and increasing pressure. Shearing is achieved by trapping the sludge between two belts and passing them over a succession of rollers, so that each belt takes up the inner and outer position alternately. Since the linear speed of the belt is constant, and the outer path is greater than the inner path, a difference in angular velocity is brought about. The importance of the shearing action merits particular attention. Gradual particle realignment brought about by the shear movement minimises interstitial space and is the most important mechanism of separation by the belt filter press.

The cake produced from the very fine slurries is of such a dryness that it is capable of being returned to the discard tip, or used for reclamation purposes, resulting in considerable cost savings when dressing the tip for final landscaping(32). The filtrate is of sufficient quality to enable reprocessing of the water for further washing of the slurries.

7. CONCLUSION

Increasing demands on natural resources, the limited availability of fresh water, and rising operational costs have focused on the need for conservatism. The utilisation of recycled water, while contributing to resource conservation introduces operational problems in terms of suspended solids retention and increasing dissolved solids concentrations. The technology for effective, energy efficient purification and optimisation of recycled water exists, and as the demand for stricter resource management and process efficiencies develop, these technologies will be harnessed and refined. Filtration will be an essential component in mine water recycling and reclamation.

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DISINFECTION

1. INTRODUCTION

Underground mine service water in South Africa is used for dust suppression, drilling, cooling and cleaning. Although the chilled water is not intended for potable use, it is often used as drinking water by mining personnel and atomised droplets may be inhaled. The water is recirculated within the mining circuits where it may be infected with intestinal organisms or water borne diseases such as cholera or typhoid. In order to protect the health of the underground work force it is customary to disinfect the service water.

Although a regulation is in force that all water used underground shall be free of bacterial contamination, in practice it is often very difficult to accomplish this (2).

Industry, utilities and municipalities have relied on chlorine gas as the principal method for microbiological control of cooling waters and disinfection of waste treatment applications. Today the concerns for the environment, operator safety, and liability issues have caused many plants to reconsider their use of chlorine and look at alternative biocides that meet environmental discharge requirements and minimise safety and liability issues. Bromine chemistry has proven to be a simple, cost effective, alternative to chlorination/dechlorination for these applications.

Chlorine gas cannot be used underground for safety reasons and chlorine for safety reasons and chlorine compounds such as calcium or sodium hypochlorite have traditionally been used.

2. PATHOGENS

Disinfection refers to the selective destruction of disease-causing organisms as opposed to sterilisation, which is the destruction of all organisms.

The bacteriological quality of the water is monitored by enumerating total aerobic bacteria (TAB), total coliforms (CC) and *Escherichia coli (E. coli)*. *E. coli* is an indicator of human faecal contamination(13). The bacteriological standard specified by Anglo American Corporation for mine service water is less than 3 000 TAB per 100 mℓ, less than 2 CC per mℓ and the total absence of *E. coli* (1).

The pathogens can be divided into different groups of organisms(8) and more detail on this is given in the literature review on microbiological contamination.

There are several disinfecting agents available for use. Among these are the use of UV irradiation, ozone, micro-filtration, chlorine dioxide and compounds that produce hypobromous and hypochlorous acids.

3. CHLORINE BASED DISINFECTION

Historically the methods of treatment have been chlorine based and include *the use of gaseous chlorine (surface installations only), calcium hypochlorite* granules or tablets, chlorinated cyanurate tablets, sodium hypochlorite solutions and hypochlorite produced by electrolysis. These compounds all produce hypochlorous acid which is the primary sanitiser in all chlorination procedures(1).

The standard method of disinfection in South African mines is treatment with calcium hypochlorite, although liquid sodium hypochlorite is used at a few sites(2). Gaseous chlorine is invariably considered to be so toxic and hazardous that chlorine cylinders are not taken underground. Solid or liquid chemicals that are easier to transport and safer to handle are used instead. Several chemicals such as sodium hypochlorite, chloride-of-lime, lithium hypochlorite, and granular calcium hypochlorite can be utilised.

Important variables affecting the efficiency of chlorine disinfection are: contact time; initial mixing; pH; temperature; concentration of disinfectant; concentration of ammonia; types of organisms; and nature of suspending liquid.

3.1 EFFECT OF CONTACT TIME

Effect of contact time is indicated by the following equation:

$N_t / N_o = kt^m$

N_o = number of organisms at time $t = 0$
N_t = number of organisms at time t
t = time (sec)
k = constant (sec^{-1})
m = constant

The equation results from the analysis of chlorination data that have been found to plot as straight lines on log-log paper.

3.2 EFFECT OF INITIAL MIXING

It has been shown that the addition of chlorine in a highly turbulent regime (Re > 10 000) resulted in kills two orders of magnitude greater than for addition under low turbulence.

3.3 EFFECT OF pH

The common chlorine compounds form hypochlorous acid on reaction with water. The hypochlorous acid (HOCl) is in equilibrium with the hypochlorite (OCl^-) ion, and the equilibrium changes with pH as shown in the Table below.

pH	5	6	7	8	9	10
%HOCl	100	93	70	22	4	0
%OCl^-	0	7	30	78	96	100

The pH is important because the HOCl has a disinfection power 40 - 80 times higher than OCl^-.

3.4 EFFECT OF TEMPERATURE

The effect of temperature on rate of kill is given by the van't Hoff-Arrenhius relationship as follows:

$$\ln t_1 / t_2 \quad = \quad (E(T_2 - T_1) / (R.T_1.T_2)$$

E = activation energy (J/mol)
t_1, t_2 = time for given percentage kill at temperatures T_1 and T_2 respectively
T_1, T_2 = temperature (°K)
R = gas constant (8,314 J/mol.°K)

where the activation energy E varies for different chlorine compounds at different pH values as shown in the Table below. Increased temperature results in a more rapid kill.

Parameter	Chlorine			Chloramines		
pH	7,0	8,5	9,8	7,0	8,5	9,5
E (J/mol)	34 332	26 796	50 242	50 242	58 615	83 736

3.5 EFFECT OF DISINFECTANT CONCENTRATION

Depending on the type of chemical agent, it has been observed that, within limits, *disinfection effectiveness is related to concentration by the following* empirical relationship:

$$C^n.t_p = \text{constant}$$

C = concentration of disinfectant
^{n} = constant
t_p = time required to effect a constant percentage kill

The constants in this equation can be determined by plotting on log-log paper, the concentration versus time required to achieve a given percentage kill. In general, if n is greater than 1, contact time is more important than dosage; if n = 1, the effect of time and dosage are about the same.

3.6 EFFECT OF AMMONIA CONCENTRATION

Ammonia reacts rapidly with hypochlorous acid to form monochloramine (NH_2Cl) and dichloramine ($NHCl_2$). An excess of chlorine will form nitrogen trichloride (NCl_3). The dichloramine is about 120 - 150 times less effective than hypochlorous acid, while the monochloramine is about 1 000 times less effective for killing bacteria. Relative ratios of mono- to dichloramine are dependent on pH, temperature, contact time and initial concentration of chlorine. Dichloramine predominates at pH 5 and below and monochloramine predominates at pH 7 and above.

Free chlorine can only be obtained once all the ammonia has been oxidised to chloramines. This is referred to as breakpoint chlorination and requires 8 - 10 mg/ℓ chlorine per mg/ℓ ammonia. Chloramines are just as effective in disinfecting water, provided sufficient contact time is available

3.7 TYPES OF ORGANISMS

Effectiveness of disinfectants is influenced by the nature and condition of the micro-organisms. Viable growing bacteria cells are killed easily, while bacterial spores are extremely resistant. Chlorination is less effective for viruses than for bacteria. In general high free chlorine residuals are required to inactivate viruses.

3.8 NATURE OF SUSPENDING LIQUID

The nature of the liquid has an important effect on disinfection efficiency. Ammonia will react with chlorine to produce the less effective chloramines. Turbidity will reduce the effectiveness of disinfectants by absorption and by protecting entrapped bacteria.

It has been shown in the past sixteen years, particularly in warmer climates, that there are disease-producing pathogenic organisms in the service water (*Salmonella* strains, *Entamoeba hystolytica* etc.), which are more resistant to chlorination than *E. coli*. Hence, the absence of *E. coli* after chlorination is not always an indication that all potentially harmful pathogens have been eliminated.

In mine service water, it is important that disinfectant maintains an active residual in the water to prevent the growth of bacterial slimes in the distribution network and refrigeration units (5).

On site chlorine generators are available on the market, and the production of chlorine by electrolysis of a saturated brine solution is an established method being used.

Chlorine generation is achieved as follows;

Dissociation	$2NaCl -> 2Na^+ + 2Cl^-$
Anodic reaction	$2Cl^- -> Cl_2 + 2e^-$
Cathodic reaction	$2H_2O + 2e^- -> H_2 + 2OH^-$
Reassociation	$2Na^+ + 2OH^- -> 2NaOH$.
Chlorine dioxide manufacturing	$2NaClO_2 + Cl_2 -> 2ClO_2 + 2NaCl$

Chlorine is produced at atmospheric pressure, making the plant safer, and only salt is stored as chlorine is produced on demand.

For a considerable number of years, kits for the in situ electrical generation of chlorine water (in the form of sodium hypochlorite) have been available commercially.

These units are not practical for industrial use since the carbon electrodes have an active life of only about three to six months.

Early in 1976 it was learnt that industrial units with platinized titanium electrodes, guaranteed for a minimum life of five years, were being manufactured to produce chlorine from sea water. These chlorinators consist essentially of a transformer that supplies direct current to platinized titanium anodes and titanium cathodes arranged in the form of tubes. The chlorine is generated as sodium hypochlorite during a 'once through' passage of sea water through the tube cell(2).

The system appeared to have considerable potential if it could be adapted for underground use. It was realised that in these circumstances, a manufactured brine solution would have to be recirculated relatively rapidly through the chlorinator while a slight controlled discharge from the circuit into the mine service water was allowed.

The assembly has sophisticated controls to prevent overheating , and has cut-outs for excess currents and low liquid flow, a leak detector and automatic reset and level controls switches. Its power supply is designed to convert a standard three-phase 50/60 Hz supply into the high current, low-voltage power required to perform this function as follows: a step-down transformer, a saturable reactor assembly, a variable controller, an auxiliary transformer, and a rectifier assembly. These components are housed in a lockable power-supply enclosure.

The electrolysis of salt water releases hydrogen at the rate of 0,25 m^3/kg of sodium hypochlorite generated. The hydrogen remains entrapped in the pipe work and is released into the atmosphere over the recycle tank. Measures should be taken to provide continues forced-draught ventilation at the location to sweep away the hydrogen. This draught also assists in cooling the circulation brine.

To ensure that all, or as much as possible of the underground service water is chlorinated, the site of installation must be carefully chosen. In most cases an appropriate site is either into, or at the outlet from, the dam feeding the machine water to the working levels below. Four units were tested at Western Holdings, President Brand, Vaal Reefs and Western Deep Levels.

The ideal characteristics of a disinfectant are compared with actual characteristics of some common chlorine-based disinfectants in *Table 1* below.

Table1: Ideal and actual characteristics of chlorine-based disinfectants

Characteristic	**Ideal**	**Chlorine gas**	**Sodium hypochlorite**	**Calcium hypochlorite**	**Chlorine dioxide**
Toxicity to micro-organisms	Highly toxic at low conc.	High	High	High	High
Toxicity to man	Non-toxic	Highly toxic	Toxic	Toxic	Toxic
Solubility	Soluble in water	Slight	High	High	High
Stability	Effective after kept in storage	Stable	Unstable	Stable	Unstable - generate on site
Interaction with ammonia	Should not interact	Strong interaction	Strong interaction	Strong interaction	No interaction
Penetration	Must penetrate through scale	High	High	High	High
Corrosivity	Should not corrode metals	Highly corrosive	Corrosive	Corrosive	Highly corrosive
Availability	Available in large quantities and low cost	Low cost	Moderately low cost	Moderately low cost	Moderate cost
Handling	Safe to handle	High hazard	Medium hazard	Medium hazard	High hazard

4. CHLORINE DIOXIDE-BASED DISINFECTION

The effectiveness of ClO_2 to disinfect poliovirus and indicator bacteria in clean water and sewage was determined during various studies. *Clostridium perfringes* was the most resistant to disinfection by ClO_2, followed by poliovirus, *S. faecalis* (faecal streptococcus) and *E. coli*. The results suggest that reduction in concentrations of *C. perfringens*, but not faecal coliform or *faecal streptococcus can be used as a simple monitoring assay to determine* if ClO_2 treatment conditions to disinfect human enteric viruses had been achieved. Some resistance to disinfection has been detected and is usually associated with suspended matter that hinders disinfection. Settling of mine service water should eliminate this problem, see literature review on flocculation and settling.

Chlorine continues to be the disinfectant of choice to treat both drinking and waste waters. However the USEPA (1976) task force report concluded that the standard practice of treating waste waters with chlorine resulted in the formation of toxic as well as potentially carcinogenic products and was also ineffective in disinfecting human enteric viruses. Of the alternative disinfectants tested, chlorine dioxide is one of the most promising since:

- it does not react with hydrocarbons to form carcinogens;
- remains an effective disinfectant even in the presence of ammonia and organics in water;
- remains a residual in the water (14).

The presence of organic nitrogen and ammonia in underground mine water often results in inadequate disinfection being achieved. A case study at a mine on the Witwatersrand Reef indicated that chlorine dioxide could be an economically viable alternative to chlorine, especially for underground applications. Disadvantages of using this disinfectant are that sodium chlorite used for its generation has to be imported from overseas, and a somewhat more sophisticated generation-dosing system than chlorine(5).

Minimizing the inorganic by-products, chlorite ion and chlorate ion, in drinking water may be important if chlorine dioxide is used for potable water treatment. Sulphur dioxide, sulphite and free chlorine are used to do this in America as the USEPA has limits on the concentrations of these ions in drinking water.

5. BROMINE BASED DISINFECTION

Bromide can be delivered for water treatment applications in three practical forms:

1. *Bromochlorodimethylhydantoin* (BCDMH) (solid tablets or granules)

   Reaction: $C_5H_6BrClN_2O_2 + 2H_20 \rightarrow C_5H_8N_2O_2 + HOBr + HOCl$

   Feed system: Brominator

2. *Activated Bromide* (Br-) (liquid)

   Reaction: $HOCl + Br^- \rightarrow HOBr + Cl^-$

   Feed system: Positive displacement pump

3. *Bromine Chloride* (BrCl) (liquid in 3 000 lb cylinders or bulk)

Reaction: $BrCl + H_2O \rightarrow HOBr + H^+ + Cl^-$

Feed system: Bromine Chloride Feeder

The reactions above show how bromine, in its various forms, hydrolyses in water to form the weak acid, hypobromous acid (HOBr). Regardless of the form in which bromine chemistry is delivered, the principal mechanism for biocidal activity is the formation of hypochlorous and hypobromous acid in the chlorination of water. Both hypohalous acids are powerful oxidising agents which combine readily with protoplasm in an organism forming stable *nitrogen-halogen bonds with the proteins. The formation of these halogen bonds interrupts the metabolic processes and is, therefor, toxic to all living* organisms(3).

Since both bromine and chlorine are in the same chemical family known as halogens, there are many similarities in how the two elements react in water treatment applications. However, there are some key distinctions.

5.1 NITROGEN ENVIRONMENT

Since various levels of ammonia and/or nitrogen are encountered in most mine water (due to explosive residue), it is important to consider the effect of these substances on biocidal activity and final effluent residuals of bromine and chlorine in treated systems.

Halomines are quickly formed when the biocidal active agent, hypohalous acid (HOBr or HOCl), comes in contact with nitrogen based compounds.

Chloramines are formed upon chlorination and are relatively poor biocides. Monochloramine is predominant at the pH of typical water treatment applications. At a pH of 8,5 monochloramine formation is virtually complete. The biocidal activity of monochloramine is eighty times less than that of free chlorine(3).

Mono- and dibromamines are formed during bromination, and are very active biocides. These bromamines show disinfection properties that are comparable to that of free bromine. In some systems, breakpoint chlorination is required to maintain microbiological control when ammonia is present. *Typically breakpoint chlorination requires 10 mg/ℓ of chlorine for every 1 mg/ℓ* of ammonia. Since bromamines are very effective biocides, breakpoint bromination is not relevant.

In summary, effective microbiological control and disinfection can be achieved at much lower halogen dosages when bromine chemistry is applied.(10)

Alternative halogen-based disinfection methods were studied and a dramatic effect of a low level nitrogen (2 mg/ℓ) on biocidal activity of bromine and chlorine chemistries against three bacterial genera. Chlorine was effective against *E. coli* but was less effective against the *Pseudomonas*, and *Streptococcus* genera. Pseudomonas is known for its relative resistance to biocides and is the predominant slime forming bacterial genus of concern in condenser and heat exchanger biofouling.

5.2 KINETICS

The kinetics of halogen-based disinfection methods were also studied and bromine chemistry achieved kill levels of at least four orders of magnitude in less than four minutes. These results take on particular significance in applications that require rapid disinfection. The disinfection kinetics of bromine chemistry make it possible to reduce contact time in these applications. This reduction in exposure not only benefits the environment, but can result in lower condenser corrosion rates.

5.3 ENVIRONMENT

An analysis of the relative decay ratios of bromamines vs. chloramines helps explain why bromine chemistry has a positive impact on the environment. Bromamines decayed to low levels in less than half an hour while chloramines required many hours to decay to the same levels.

Several fish survival studies have confirmed that the environmental impact is reduced significantly through the application of bromine chemistry(4).

5.4 ALKALINE WATERS

With the decline of chromate treatments in recirculating cooling systems, there has been a trend toward alkaline corrosion control programs. It has been well documented that the effectiveness of chlorine is reduced significantly in an alkaline environment. As the pH of the water rises, hypochlorous acid (HOCl) and hypobromous acid (HOBr) dissociate to form the hypohalite ions (OCl- and OBr-). The hypohalite ions are not active biocides at typical use levels. Disinfection performance in an alkaline environment shows that bromine chemistries continued to achieve kill levels of 4 to 6 orders of magnitude at a pH of 8,2.

In mine service water systems, the effects of fluctuating pH values and high levels of ammonia and suspended solids create many problems with chlorine based disinfection compounds. Many mines have service water systems with pH values above 7,5 - 8, due to poor neutralisation control, and ammonia levels as high as 100 mg/ℓ or more. Under these conditions, effective chlorination is almost impossible.

Over the last few years a number of mines have evaluated and implemented bromine based disinfection systems, using the chemical bromo-chloro-dimethyl-hydantoin which releases both active chlorine and bromine compounds. The bromine portion remains an active and effective disinfectant at much higher pH values than chlorine as shown in the Table below.

pH	Chlorine		Bromine	
	% HOCl	% OCl	% HOBr	%OBr
7,5	50	50	100	0
8,5	10	90	60	40

NOTE : Both HOCl and HOBr are almost 2 orders of magnitude more effective as disinfectants than OCl and OBr.

5.5 FIELD TRIALS

Trials using BCDMH were conducted at two shaft sites, one in the Free State and one in the Western Transvaal (now North West Province). The BCDMH was used in tablet form and was dosed by means of a dispenser which had a 35 kg capacity.

Results from both trials showed that the BCDMH had been successful in maintaining low bacterial counts. Cost comparisons were calculated for both HTH and BCDMH in both trials.

A 41% cost saving was calculated for the trial run in the Free State. However in order to ensure proper biological control, it has been agreed that the dosage rate be increased to 1 mg/ℓ, which would still represent a cost saving of 34,4%.

The cost of the trial run in the Western Transvaal Region was calculated at an average monthly cost of 29% saving.

6. OTHER DISINFECTION TECHNIQUES

6.1 OZONE BASED DISINFECTION

Inactivation data with ozone in a system shows high reactivity and strong biocidal potential. Disinfection with ozone will be more efficient at higher ozone residual within a shorter contact time(19).

Ozonation is a relatively new means of water disinfection, and according to the regulations issued by the U.S. Dept. of Labour, Occupational Safety and Health Administration, the maximum permissible concentration in drinking water is 0,1 mg/ℓ. The problem for underground water is that the residual ozone is not as effective in preventing regrowth and recontamination.

Equipment that produce ozone, usually require ducts that remove the exhaust air, this makes it impractical for underground use. Furthermore the ozone production process is complicated and more expensive than other methods.

6.2 ULTRAVIOLET IRRADIATION

Ultraviolet has been known to have biocidal properties. A system that provides equipment for thin film application, as indicated by independent tests, is effective for potable water treatment as well as waste water effluent disinfection.

Methods of photochemical disinfection for crop irrigation have been devised, using sunlight as UV (ultraviolet) source and dissolved oxygen (DO) as the oxidising agent and methylene blue (MB) as an intermediary for the absorption and transfer of sunlight energy to active DO and/or to destabilise the organic matter(7).

Human enteroviruses, predominantly poliovirus type 1, have been found to survive disinfection consisting of a nominal 1,0 mg/ℓ of free residual chlorine maintained for a minimum contact time of 15 minutes at a pH value of 7,2 and a temperature of 11 °C. Disinfection with 1,5 mg/ℓ of chlorine under the same conditions resulted in virus inactivation and they were also destroyed by exposure to 25 mJ/cm^2 of ultraviolet light(14).

Ultraviolet radiation is usually applied by lights that need maintenance and often cleaning. The equipment used is not suitable for underground use and the water often has a tendency towards scaling, and may also have a high turbidity. Turbidity scatters and absorbs ultraviolet rays and the suspended particles may also contain pathogens in them, which will not be killed. One of the major factors that makes ultraviolet irradiation unsuitable for mine water treatment is that it has no residual disinfection potential. In a mine water system, there are storage dams and other places in the circuit where the water may be contaminated again after disinfection and for this reason, it is important that the disinfection process used must have the ability to prevent regrowth or contamination after initial disinfection.

6.3 MATURATION PONDS

Although not practical for mine water use in terms of flow rates and surface area, maturation ponds may be used for disinfection of contaminated water.

Maturation ponds of 10 days median retention time were found to successfully disinfect a poor quality trickling filter effluent and were effective at reducing virus densities(9).

6.4 METALS AND FLOCCULATION

Silver is another substance that is known for its disinfection abilities. Silver is reportedly effective against a wide range of bacteria and viruses, has few if any after-effects, and has long lasting pathogen killing properties. A chlorine-free preparation for the disinfection of containers, pipes, and other equipment. It is available in powder or tablet form and an electrolysis system using silver anodes to treat large quantities of water(6).

Copper sulphate is also used in some cases, to control the growth of algae and other micro-organisms in reservoirs and water bodies(6).

These methods are not used for disinfecting mine service water as the metals have residual toxicity for workers drinking the water and these techniques are expensive.

Studies show that some pathogens like Hepatitis A and Rotavirus are removed from water by flocculation, coagulation, filtration and softening to a certain extent. Coagulation, settling and filtration in a pilot plant lowered the densities of these pathogens by 98,4 - 99,7 percent(15).

7. POTENTIAL PROBLEMS ASSOCIATED WITH DISINFECTION

Primarily because the standard method of disinfection depends on human control for the addition of hypochlorite and the regulation of its flow, it is frequently found difficult to disinfect mine water adequately and continuously(2).

Gaseous chlorine is invariably considered to be so toxic and hazardous that chlorine cylinders are not taken underground. Although widely employed, calcium hypochlorite is not the ideal additive for use underground. One of the main disadvantages is the difficulty in feeding the solid, or a relatively concentrated solution of it, at a constant dosage rate. Apart from chlorine gas, it is the cheapest form of available chlorine but it is still expensive to use. The hypochlorite increases the pH value of the service water, which in practice decreases the disinfecting activity of the hypochlorous acid solution formed; additional calcium is added to the water, increasing its scaling tendencies, which are probably already high(2).

The optimum procedure for effective disinfection requires sufficient contact time between micro-organisms and disinfectant to allow for proper kill, followed by a level of residual biocide to prevent bacterial regrowth.

It is often difficult to maintain the optimum procedure as factors such as fluctuating pH, the presence of ammonia and a high level of organics in water all interfere with the biocidal action of hypochlorous acid(1).

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DESALINATION

1. INTRODUCTION

Desalination is a concentrating process whereby a saline feed water is separated into a more concentrated brine stream and a relatively pure water stream. The main application is in the recovery of fresh water from sea water using various distillation techniques. Sea water desalination has several advantages over the typical mine water treatment. Since the supply of sea water is virtually unlimited it is possible to operate these desalination processes at low recovery rates. The composition of the saline feed is fairly constant and the disposal of the brine product does not present particular difficulties.

It is becoming increasingly beneficial and even essential for mines to recycle as much of their service water as possible in trying to achieve an ideal closed circuit and thereby reduce the amount of often costly make-up water which is required. The value of desalination, from an operating cost viewpoint has already been demonstrated for gold mines in South Africa (9). Often the mine has a source of good quality fissure water which has a dilution effect on the recycled service water but generally the salinity of this water reaches unacceptable levels. Therefore, the main mining application of desalination is the reduction in salinity of the recycled service water to reduce problems such as scaling and corrosion of pipes and other process equipment. Since the background philosophy of the mine water desalination process is one of water conservation it is obvious that maximum recovery from the desalination process is desirable. The volume of waste water rejected needs to be minimised and this implies that the concentration of the brine needs to be as high as possible. It is this factor which dictates which desalination process should be used.

Mine waters can be classified as either scaling or non-scaling. Scaling waters present problems when, due to the concentration of the brine stream, salts such as calcium sulphate exceed their solubility limits and precipitation occurs on heat transfer and membrane surfaces. This scaling problem eliminates many conventional desalination processes as an option. In particular, distillation processes with their associated high operating temperatures, are often not suitable for desalinating scaling type waters. In South Africa it is estimated that about 75 % (1) of gold mines have scaling type waters and it is essential that a cost effective desalination process be developed for this particular problem.

2. MEMBRANE PROCESSES

2.1 REVERSE OSMOSIS (RO)

When a volume of pure water and a volume of a saline solution are separated by certain semi-permeable membranes there is a flow of water through the membrane by osmosis into the saline solution creating a hydrostatic head of solution which at equilibrium equals the osmotic pressure of the saline solution. In reverse osmosis an external pressure, greater than the osmotic pressure, is applied to the saline solution and the flow of pure water is reversed through the membrane with the effect of concentrating the saline solution and producing relatively pure water. The viability of this process has been very closely linked to the development of membranes which combine high salt rejections, high water fluxes and mechanical strength to withstand operating pressures of up to 8000 kPa. Cellulose acetate, the first industrial RO membrane, combined these properties as well as the ability to be easily moulded into thin films. The need for mechanical support of these thin membranes has resulted in the development of various membrane configurations which include flat, tubular, spirally wound and hollow fibres.

A major disadvantage of the RO process is the precipitation of certain salts when their respective solubility limits are exceeded as the brine stream becomes more concentrated. This leads to fouling of the membrane which inhibits permeability as well as other associated scaling problems. Potential fouling problems and deterioration of cellulose acetate membranes are discussed by various authors (2, 3, 4, 5). To eliminate this problem expensive pre-treatment of the feed is required. Normally this pretreatment involves acidification to minimise membrane hydrolysis and/or softening to reduce the calcium content of the feed water.

Other fouling that can occur on membranes is the precipitation of metallic oxides, particularly ferric hydroxide which occurs when the ferrous ion is oxidised to the ferric ion as the water passes through the RO process. Suspended particles are also liable to clog the membranes as they act as a mechanical filter. Colloidal fouling is a particularly difficult operating problem since it is difficult to predict the colloid content of a proposed feed water. It is possible to eliminate fouling problems with the correct feed pretreatment but this often makes the RO process prohibitively expensive and other options need to be investigated.

The pilot plant test facility of a RO process followed by seeded distillation, crystallisation and salt drying is described by Sikora et al (6). The feed water is mine effluent from a coal mine in Poland, typically high in sodium chloride content. The pretreatment of the feed to the spiral wound membranes was critical and proved to be the most problematic stage of the process with the result that further pilot plant test work was recommended to establish the effect of the feed water quality variability. A pilot plant study on the desalination of mine service water containing high levels of dissolved sodium and chloride using tubular reverse osmosis was carried out by Juby and Pulles(7). Initial membrane fouling problems were addressed by the removal of iron via oxidation, flocculation and filtration pretreatment. The plant was operated successfully thereafter and an average salt rejection of 82 % was achieved. The conceptual design and cost estimation of a full-scale plant, operating at 85 % product water recovery, are also discussed.

The results of pilot plant studies may be used in conjunction with a TRO model developed by Brouckaert[8] to enhance the value of such information and as a prediction tool for process scale up.

2.1.1 Seeded Reverse Osmosis (SRO)

In a seeded reverse osmosis process the expensive pretreatment requirements are greatly reduced by actively promoting precipitation of scaling species onto preferential sites instead of on membrane walls. The idea of calcium sulphate crystal seeding was first introduced in the field of distillation but was developed by Resources Conservation Company (RCC) for use in the reverse osmosis application (10).

The pilot scale operation of an SRO plant treating mine service water is described by Harries (11). The preferential precipitation sites are a suspension of seed crystals which are introduced into the feed via a recycle from the reject stream. At start-up the crystals need to be added as an initial batch dosage. The slurry of seed crystals is approximately 3-10% solid content and a tubular reverse osmosis (TRO) configuration is required because spirally wound or hollow fibre configurations are unsuitable for such concentrations of suspended solids. The saline water is fed into a recirculation tank where it is seeded with the calcium sulphate crystals. After pressurising to about 4 MPa the slurry is fed into the bank of TRO modules. As the permeate is extracted and the brine becomes more concentrated the solubility limit of calcium sulphate is exceeded and precipitation occurs on the seed crystals. This reject slurry is then throttled down and sent to a hydrocyclone from where the overflow is blown down as waste and the solid seed material in the under flow is delivered to the recirculation tank to be mixed with the saline mine feed water. The 5 m^3 pilot plant was run for 5000 hours and produced a water recovery of 92-96 %. The required pretreatment was reduced to a coarse particle filtration, pH control for membrane protection and chlorine addition to control biological growth.

2.1.2 Slurry Precipitation and Recycle Reverse Osmosis (SPARRO)

Although SRO plants showed reductions in pretreatment costs, high salt rejections and high water recoveries, several disadvantages were also apparent from pilot plant studies. These included high energy consumption, high $CaSO_4$ recirculation rates and poor control of $CaSO_4$ seed and brine flows. These problems were addressed by development work undertaken by COMRO and resulted in the patent of the SPARRO process in 1988 after extensive pilot plant work which is described by Pulles et al (12).

The pretreatment for the pilot plant studies consisted of increasing feed pH to approximately 10, the removal of metal precipitates, cooling, filtration and readjustment of the pH to between 5-6 for membrane protection.

The results of the extensive pilot plant operation showed that a water recovery of 95 % was achievable. The other important parameter in terms of operation is the membrane flux rate which is a function of temperature and pressure. Problems maintaining a minimum flux rate of 550 l/m^2.d were experienced due to membrane fouling. It was identified that the fouling was probably due to the high levels of quartzite suspended material and the possibility of encapsulating these particles in the growing calcium sulphate seed crystals would eliminate this problem.

The findings from the pilot plant study were used in the conceptual design and costing of a full-scale plant to be operated at 90 % water recovery. This cost was compared to updated estimates of similar capacity electrodialysis reversal (EDR) and TRO plants and the comparison showed that although capital costs of the SPARRO plant were higher the operating costs were substantially lower, 31 % and 25 % respectively.

2.1.3 Low Pressure Reverse Osmosis

The inherent advantages in reducing the working pressure of a reverse osmosis process are obvious and include lower capital costs for membrane containment vessels and fittings as well as reduced operating costs as regards power consumption and equipment wear and tear. Bench-scale tests were carried out on a process operating at a pressure of 600 kPa and a water recovery of 80 % was achieved. The investigative work is described by Everett and Gussmann (13).

In order to increase the expected life of the membranes, which can form a substantial portion of operating costs for any membrane process, it was necessary to include several pretreatment steps. Chlorination of the feed water reduced the bacterial count and prevented blinding of the membrane surface but it would then be necessary to remove any residual chlorine as this is also detrimental to RO membranes. The softening of the water to prevent the formation of calcium sulphate scale on the membranes was investigated and the best option was found to be ion exchange on zeogel, an inorganic compound.

It was suggested that this process could be combined with a high pressure system which would treat the small volume of brine solution produced and the overall recovery would be then increased to 94-95 %. A pilot plant, incorporating only the low pressure RO and the required pretreatment, was installed at Western Areas Gold Mine but it is the costly softening requirements for this process which make it uneconomical for any scaling-type mine water application.

2.2 ELECTRODIALYSIS AND ELECTRODIALYSIS REVERSAL (EDR)

This process uses direct electrical current applied across a stack of alternating anion- and cation- selective membranes. The anions in the raw feed are pulled towards the anode but are unable to pass through the cation-permeable membrane and thus become held or concentrated in a concentration chamber. The cations, which are moving in the opposite direction, are impeded by the anion-permeable membrane and are concentrated in the same chamber. The chambers which the ions leave become dilution chambers from which the desalinated product water is extracted.

This process was greatly improved in 1970 when the concept of current reversal was introduced(14). The anode and cathode are periodically interchanged as are the brine and product water flow channels. This periodic polarity reversal facilitates membrane cleaning and reduces the potential for membrane scaling thus eliminating the need for extensive process pretreatment. After reversal the chambers are flushed to remove colloidal material, scale products and other suspended matter. The only pretreatment that is required is a fairly fine filtration.

Apart from the reduction in pretreatment requirements electrodialysis reversal has other advantages over reverse osmosis processes. EDR membranes can generally operate over a wider pH and temperature range. The construction and maintenance costs are reduced due to the lower working pressures possible and the membranes are also more accessible and stable. In addition the membranes do not suffer the gradual compaction which is the case in the high pressure reverse osmosis processes. Problems of membrane fouling were, however, experienced at the EDR plant at ERPM and these were attributed to calcium sulphate scaling and inadequate pretreatment (15).

An investigation of EDR at pilot plant scale was carried out by Juby and Pulles(16) on service water at Beatrix Gold Mine. This water is high in dissolved sodium and chloride and also contains iron and manganese which is removed by oxidation and filtration pretreatment. The pilot plant achieved a product water recovery of 84 % and a average salt rejection of 80%. The conceptual design and costing of a full-scale plant was carried out based on the findings of the investigation.

3. NON-MEMBRANE PROCESSES

3.1 ION EXCHANGE

Ion exchange is defined as the reversible exchange of ions between a solid and liquid and in which there is no substantial change in the structure of the solid, which is usually a type of resin. The process of ion exchange may be used to remove potential scale forming ions as a pretreatment for other desalination processes(17) or as a stand alone desalination technique, although this option is usually limited to water with salinity values of 3000 mg/l.

The De Sal process(18) uses an anion exchanger with intermediate or weak basicity and a carboxylic cation exchanger. The regeneration of the respective resins is achieved with ammonia and sulphuric acid. A variation of the De Sal process has been used to treat acid mine drainage to a potable quality in a 3Ml/day facility at Philipsburg, Pennsylvania(18).

Ion exchange can be used in a variety of bed configurations and with various resin types but the regeneration of the loaded resin is a critical component of the process and the required reagents are often prohibitively expensive.

3.1.1 Ion Exchange in Fluidized Bed (GYP-CIX)

The registered GYP-CIX process uses the mechanism of ion exchange to remove cations and anions in separate loading sections from saline mine effluent streams which are high in dissolved calcium and sulphate. The principles of a pilot plant operation are discussed by Gussmann et al(19).

The saline feed water is fed counter-current through a fluidized bed of air-lifted resin. The loading section may be a simple vertical loading column or a multi-stage cascade configuration. The cation exchange occurs first and the fully loaded resin is discharged into a batch regeneration vessel at the end of the cation loading. Regeneration of the resin is achieved by the addition of a 10 % sulphuric acid solution and the resin is returned to the start of the loading section. After decarbonising, the water passes through the anion exchange section. Resin from this section is regenerated by the addition of a 0.5 % lime solution to the regeneration vessel. In both regeneration sections, the sulphuric acid and lime solutions are seeded with gypsum crystals to enhance the precipitation process. Precipitation onto, and thus fouling of, the resin beads is avoided due to the charge repulsion which exists between functional groups on the resin and the co-ions in solution. The product from both the regeneration systems is a solid gypsum precipitate which is settled out and is then more easily disposed of than the brine product from other desalination processes.

The literature describes the 24 m^3/day pilot plant operation which yielded water recovery rates of >95 % and showed 97 % reductions in calcium and sulphate ion concentrations. It was also found that radio-active elements such as uranium and radium were removed and reported to the solid gypsum waste. Cost estimates for the pilot plant operation are given but the conceptual design and costing of a larger scale facility are not described.

3.2 FREEZE DESALINATION

Since most underground mines require some degree of cooling at deeper levels, the process of freeze desalination has been investigated as a means of obtaining a relatively pure, chilled water underground(20, 21, 22). The desalination is achieved by the formation of pure ice crystals as freezing occurs and the high purity ice slurry is then sent underground to be used as service water where the increased cooling effect of melting ice is achieved.

The choice of process for the ice-making depends on whether the cooling ability or the water purity is the overriding requirement. If cooling is the most important requirement then conventional ice-makers may be used but special freeze desalinators would be required if a high purity product was required. The limitation in conventional ice-makers arises from the freezing point depression as the salinity in the recirculating brine increases. A high recovery ratio requires the recirculation salinity to be very high which corresponds to a lowered freezing point. A further drawback is the tendency for salts, which are concentrated near the ice-water interface, to become trapped in the ice layer as freezing occurs.

Test results of a study by Shone and Lewis(20) were more promising for mine waters high in calcium sulphate than a typical Free State mine water which is high in sodium chloride but the problems associated with this process and the low recovery rates which are possible have limited the scope of possible applications. Frozen service water in both a solid and slurry form is used at mines such as ERPM and Vaal Reefs respectively but the motivation for these installations has been for the cooling effect achieved underground. Any degree of desalination achieved is obviously beneficial but incidental.

3.3 SULPHATE REMOVAL

The removal of sulphates from saline effluent streams may be considered as a desalination treatment because the sulphate salts are the predominant contributor to salinity in streams from mining sources in South Africa. The removal of sulphates can be achieved biologically or chemically and both methods are discussed briefly.

3.3.1 Biological Sulphate Removal

The biological reduction of sulphate to sulphide by the bacterium *Desulfovibrio desulfuricans* and the potential subsequent conversion of sulphide to sulphur has been investigated by various authors (23, 24, 25, 26, 27) as a means of sulphate removal from industrial and mine water. This process is viable for the removal of a large portion of the sulphate content and may be useful as a pretreatment step for another more complete desalination process. A useful by product of the process is the production of hydrogen sulphide which is utilised in the removal of heavy metals.

3.3.2 Chemical Sulphate Removal

Barium compounds offer the potential for sulphate removal due to the low solubility of barium sulphate but barium salts such as the hydroxide, nitrate and chloride are expensive and a recycle loop would need to be included for the regeneration of the barium content. There is the further disadvantage of adding another ion into solution which is obviously undesirable. Barium sulphide, when added to sulphate containing waters has the disadvantage of producing hydrogen sulphide. This gas has to then be stripped from the water which incurs extra cost.

The use of barium carbonate, although relatively insoluble, has the advantage of being easily recycled[28]. It can also be produced from raw materials which are available locally and it is one of the more stable forms of barium which implies that it can be conveniently handled.

The process involves the addition of a barium carbonate/lime slurry to the effluent stream in a mixed reactor with a retention time of approximately 45 minutes. Calcium carbonate seed crystals prevent scaling of the reactor vessel and enhance the precipitation which occurs via the following predominant reaction:

$BaCO_3 + CaSO_4 \longleftrightarrow BaSO_4 + CaCO_3$

The reactor product is sent to a thickener where the softened overflow may be used as a source of water for the mine or other users. The high pH of this overflow may necessitate neutralisation with sulphuric acid. The thickener under flow consisting of barium sulphate and calcium carbonate is sent to a central treatment facility after filtration and drying. It is mixed with pulverised coal and fed into a rotary kiln at approximately 1150 °C where barium sulphide and carbon dioxide are produced. The barium sulphide is then reacted with water and the kiln flue gas to produce a barium carbonate precipitate and hydrogen sulphide which, via proven technology, can produce elemental sulphur.

3.4 DISTILLATION

The effluent water from coal mines in Poland which can reach salt contents of up to 70 000 mg/l, predominantly sodium chloride, has been successfully desalinated with the use of multi-stage flash (MSF) distillation technology[29, 30].

The concentrations of the Ca and SO_4 are equalised and crystallisation on seed crystals in the brine is encouraged. This gypsum product is then separated with the use of sand filtration. The sodium chloride is removed similarly in later stages of the distillation train. The process consists of a number of flash evaporators connected in series and each evaporator is fitted with a condenser which acts as a heater for the incoming saline water.

The technology has been developed to a stable and reliable status and finds wide use in the treatment of saline waters high in sodium chloride content. The energy requirements are high although this is partly offset by the sale of the product table salt.

4. BRINE DISPOSAL

As discussed earlier, desalination is a concentrating process. The desired product is a purer, less saline effluent but the salt content removed from the feed water remains in a concentrated form and needs to be conveniently disposed of. Several methods of brine disposal exist, with evaporation and subsequent solid removal, being the most widely used.

The problem of brine disposal must be acknowledged in any evaluation of desalination processes as it may prove to be significant in terms of impact and cost.

5. CONCLUSION

The requirement and necessity for mine water desalination is increasing due to the limited availability of fresh water make-up, negative environmental impacts and increased operational costs, through scaling and corrosion problems. Conventional desalination techniques such as high temperature distillation processes are energy intensive and thus costly. In general, South African mine service waters are high in calcium and sulphate concentrations. Most desalination processes are concentrating processes whereby the salts are concentrated into a small volume of brine reject. Solubility limits are thus often exceeded, particularly at elevated temperatures, and this causes scaling of piping, process equipment and membranes. Methods to combat this scaling problem have been developed as well as processes which only remove a certain portion of the salt content, usually sulphate. Desalination is presently an expensive process and is generally not used on a large scale in the South African mining industry. As existing resources become scarcer and further development of desalination technology occurs, this situation will change.

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CYANIDE

1. INTRODUCTION

The cyanidation process for the recovery of gold was introduced into South Africa in 1890[1]. Cyanide has been used in the gold mining industry as a lixiviant i.e. a chemical which will dissolve a particular material from an ore. It is added to the crushed ore which is in a high pH aqueous suspension, in the form of sodium cyanide or calcium cyanide. The ore is leached by air agitation until all the available gold has been dissolved. The gold bearing solution is further treated to precipitate and recover the gold, it is then recycled and the spent solids are re-pulped and pumped to the tailings dam. Here the dissolved cyanides undergo natural degradation through a number of processes[2].

Usually, the tailings dam return water is reused in the metallurgical plant, however in some cases it may be pumped to evaporation dams where the potentially hazardous cyanides are oxidised or precipitated out as less toxic metal complexes. If the residence time in the dam does not allow for oxidation, ferrous cyanide is used to precipitate the cyanide out of the solution. Similarly, ferrous sulphate is used to precipitate cyanide out of the tailings which are used as backfill in mined-out stopes[2, 3]. This ensures that there is no seepage of free cyanide into the underground water system.

2. CHEMICAL AND PHYSICAL PROPERTIES OF CYANIDE

Cyanide is very susceptible to oxidation due to its carbon nitrogen triple bond. It is also thermodynamically unstable in the presence of mild oxidants,[4] however the kinetics are slow under normal conditions. A number of important reactions involving cyanide in which a multitude of both ionic and complex compounds are formed are listed below :

- Hydrocyanic acid (HCN) results from the protonation of cyanide ions (CN^-) . This reaction is highly pH dependant as shown in Table 1 below[5]. Both the CN^- and the HCN can be considered to be free cyanide since HCN is readily volatilized to the atmosphere with the rate increasing as the temperature and the turbulence increase and as the pH decreases.

Table 1 : Effect of pH on CN/HCN Ratio

pH	7	8	9	9,36(pKa)	10	11
HCN(%)	>99	96	70	50	12	<1
CN^-(%)	<1	4	30	50	88	>99

- Empty anti-bonding orbitals on the cyanide ion result in the formation of strong metal cyanide bonds[(4)]. Cyanide can form complexes with 28 different elements and 72 different metal ions. Metal cyanide compounds are less toxic than free cyanide. Iron cyanides, which are very stable may precipitate out as the characteristically blue compound - Prussian Blue.

- All metal cyanides will dissociate to release free cyanide in the presence of ultraviolet light or strong acids. The free cyanide ion will protonate to hydrocyanic acid (depending on the pH) which will volatize to the atmosphere. This reaction occurs within 3 minutes for ferrocyanide under strong UV light[(1)]. Similar reactions occur in daylight but at a slower rate. In a 'typical' tailings dam, 90 % of free cyanides will decompose over a 24 hour period.

- Sulphur compounds in solution with cyanide may react with the cyanide to form thiocyanates,[(7)] which are considerably less toxic than free cyanide. A recent proposal on the oxidation of thiocyanate has been direct oxidation from SCN to $(SCN)_3$ occurring in acidic, aqueous conditions[(4)].

- The oxidation of cyanide to cyanate occurs in the presence of strong oxidising agents such as ozone, hydrogen peroxide and hypochlorite. Cyanide is also oxidised in the presence of ultraviolet light with titanium dioxide, cadmium sulphide or zinc oxide acting as catalysts[(8)].

- Cyanide decomposes naturally due to elevated temperatures, ultraviolet radiation, low pH and bacterial activity. Decomposition involves the following processes [(5)] :

- volatilization
- biodegradation
- oxidation
- photodecomposition
- metals precipitation
- dilution
- adsorption on solids
- secondary metal precipitation

3. THE OCCURRENCE OF CYANIDE

3.1 CYANIDE-BEARING STREAMS

A sampling survey undertaken by COMRO and the mining industry in 1993 sampled various cyanide bearing streams on every gold mine in South Africa[(2)]. The samples were analysed for available cyanide (alkaline cyanides, zinc and cadmium cyanide complexes and free cyanide).

The results obtained show that the cyanide levels in tailings sent to slimes dams are highly variable (available cyanide 0,3 mg/ℓ to 114,0 mg/ℓ, average 43,0 mg/ℓ) and depend largely on the type of gold recovery process employed. The cyanide in these dams is rapidly destroyed by natural decomposition methods. The slimes dam removes about 66% of the total cyanide present in the tailings, while the toe dams and drains intercepting the seepage from the slimes dams result in a further removal of about 33% of the cyanide(2).

Another examination of gold tailings disposed of in Gauteng showed that there was little available total cyanide in interstitial waters in gold tailings impoundments (about 1 mg/ℓ cyanide) despite cyanide discharge levels from gold plants in the region of 20 mg/ℓ cyanide(8). The same writer claims that the general total cyanide levels in near surface ground water are about 0,03 mg/ℓ which shows little evidence of cyanide contamination of the ground water regime below or adjacent to gold tailings disposal facilities.

3.2 EFFLUENT WATER

Water discharged from gold mines as effluent to the surface and ground water environment consists of underground mine water (40 %), metallurgical plant water (32 %), treated sewage water (24 %) and slimes dam overflow/seepage (4 %)(9).

The total cyanide concentration of the average gold mining effluent can be estimated as underground mine water (0.17 mg/ℓ total cyanide), metallurgical plant water (0,52 mg/l total cyanide), treated sewage water (0 mg/ℓ total cyanide) and slimes dam overflow/seepage (0,46 mg/ℓ total cyanide)(2). Thus, based on a weighted average of all effluents and assuming that all the streams are mixed prior to discharge, the average gold mine effluent concentration is 0,25 mg/ℓ total cyanide which is well within the General Effluent Standard of 0,5 mg/ℓ. The average gold mine available cyanide concentration is 0,19 mg/ℓ(2).

4. TOXICITY OF MINE EFFLUENTS

According to the danger criteria defined in the HWR(10) for a typical gold mine effluent with a composition as described above mine effluents should be classified as non-hazardous waste with regard to explosivity, stability, flammability, corrosivity and chemical reactivity. The hazard assessment should concentrate on the toxicity criteria: acute toxicity, chronic health hazards, ecotoxicity, mobility, permanence and bioaccumulation(2).

4.1 ACUTE TOXICITY

The toxicity of cyanide has been shown to be influenced by hardness (calcium and magnesium concentration), alkalinity, pH, total dissolved solids, total suspended solids and many inorganic and organic complexing agents. The four important categories of cyanide in mining waste waters, from a toxicity point of view are :

- free cyanide
- iron cyanides
- weak acid dissociable (wad) cyanides
- cyanide related compounds

The main factors affecting the toxicity of free cyanide are temperature, pH, dissolved oxygen, salinity and ammonia concentration(2).

Toxicity data related to human consumption is as follows(2) :

- mean lethal dosage of free cyanide, 50 to 200 mg (relating to an LD_{50} of 1 to 3 mg/kg)(13).

- ingestion by humans of less than 10 mg/day of free cyanide is non toxic. Continuous long term consumption of up to nearly 5 mg/day has shown no injurious effects(14).

- LC_{50} for free cyanide inhalation is about 100 mg/kg of body weight.

In calculating the LD_{50} for humans, an available cyanide concentration of 0,19 mg/l is used. The worst case scenario is considered i.e. where all the available cyanide is free cyanide.(2) Using an LD_{50} of 1 mg/ℓ for humans and the worst case calculated average free cyanide concentration of 0,19 mg/ℓ the LD_{50} of the mixture is calculated using equation 1.

$$LD_{50}(\text{mixture}) = \frac{LD_{50}(\text{most toxic constituent})}{\text{concentration}(\text{same constituent})} \qquad (1)$$

$$\begin{aligned} LD_{50}(\text{mixture}) &= 1 \text{ mg/kg} \,/\, 1{,}9\times10^{-7} \text{ mg/mg} \\ &= 5{,}26\times10^{6} \text{ mg/kg} \end{aligned}$$

This value measured against the criteria in Table 2(10) indicates that such a mine effluent is non hazardous with regard to free cyanide. Alternatively a 70 kg person would have to drink 368 ℓ/day of mine effluent to obtain a toxic dose of free cyanide.

4.2. CHRONIC HEALTH HAZARDS

Chronic health hazards arise with prolonged exposure to harmful substances which have a cumulative effect and risks are therefore not reduced with dilution(2). There is no evidence that chronic exposure to free cyanide results in teragenic, mutagenic or carcinogenic effects(5). Furthermore cyanide is not accumulated in the body and is readily metabolised. Mine effluents are therefore classified as non hazardous with regard to chronic health hazards from cyanide.

Table 2 : Toxicity Rating

Danger Group	Toxicity	LD_{50}(oral) mg/kg	LD_{50}(dermal) mg/kg	LC_{50}(inhalation) mg/l
1	Extremely toxic	< 5	< 40	< 0,5
2	Highly toxic	5 - 50	40 - 200	0,5 - 2,0
3	Moderately toxic	50 - 200 (solids) 50 - 500 (liquids)	200 - 1000	2 - 10
4	Slightly toxic	500-5000	1000-20 000	10 - 100
5	Insign. toxic	> 5000	> 20 000	> 100

4.3 ECOTOXICITY

Ecotoxicity is defined as damage to natural systems and accumulation in biological food webs or persistence in the environment(2).

Trout are the most sensitive of the fish species to cyanide(5). The acute toxicity for free cyanide in fish ranges from about 0,05 - 0,1 mg/ℓ(14). Natural water bodies receiving mine effluents will generally have a pH < 8,0 therefore the more toxic form of free cyanide (HCN) will be present. Hydrocyanic acid (HCN) however, is highly volatile especially at higher temperatures in aerated mediums, this combined with the dilution effect of the receiving water ensures that the free cyanide levels will be below detection limits. The toxicity of metal complexed cyanides (and not free cyanides) is therefore of major concern(5).

Data for the toxicity of metal complexed cyanides in relation to trout has been covered at length in the literature(2). It is difficult to quantify the ecotoxicity of the average mine effluent since the breakdown of cyanide species reported as available cyanide is not known(2). If the recommended limit of 0,05 mg/ℓ free cyanide for rainbow trout is considered then the average mine effluent would be classed as ecotoxic.

It has been recommended that the determination of the ecotoxicity of mine effluents be preceded by studies which take into account various factors(2, 15):

- the standard practice of using the limits for the most sensitive species i.e. rainbow trout rarely applies under South African conditions.
- the dilution of the discharged effluent into the receiving water body.
- LC_{50} data generally only considers a single contaminant and not the complete mixture.

4.4 MOBILITY, PERMANENCE AND BIOACCUMULATION

Cyanide has a strong tendency to form metal complexes that readily precipitate, it also rapidly degrades by natural mechanisms therefore it cannot be considered mobile, permanent or bioaccumulative[2]. Mine effluents should therefore be considered non hazardous with regard to mobility, permanence and bioaccumulation.

5. CYANIDE TREATMENT

There are two methods of cyanide treatment: deactivation or recovery. Deactivation uses oxidative methods to decompose the cyanide into carbon dioxide and nitrogen. Cyanide recovery uses reagents which modify the cyanide ligand so that the product is stable.

The most well known methods of cyanide oxidation are those using chlorine, hydrogen peroxide, ozone or sulphur dioxide in the presence of air. Other oxidative methods use active carbon or convert cyanide to thiocyanates (with sulphur) or ferrocyanides (with iron salts). Cyanide may also be deactivated by biodegradation.

Techniques employed for cyanide recovery include ion exchange, chelating resins and AVR (acidification-volatisation-reneutralisation).

5.1 CHLORINE OXIDATION

The alkaline chlorination process for removing cyanide from aqueous slurries was first patented in Germany in 1943, and has been used extensively ever since for treating industrial effluents.

The chlorination proceeds via multiple step reactions which are pH sensitive[16]. Cyanogen chloride, a very toxic gas, is formed as an intermediate product (the rate of this reaction is rapid under all pH conditions), it then hydrolyses more slowly to form cyanate (the rate of this reaction is very pH dependant and increases with an increase in pH)[17]. The cyanate gradually hydrolyses to ammonia and carbon dioxide but will also react with chlorine.

Some of the complexes, such as ferro- and ferricyanide are not destroyed by chlorine while some others, such as certain copper and nickel complexes are destroyed very slowly[16]. In the presence of sulphur bearing material (e.g. pyrite) some of the cyanide can be converted to thiocyanate which is oxidised by chlorine ahead of the cyanide. This will increase the cost of using chlorine substantially. Residual chlorine in the treated effluent may require removal to meet effluent quality requirements.

5.2 HYDROGEN PEROXIDE OXIDATION

There are two hydrogen peroxide oxidation processes:

-the 'Kastone' process uses a specially stabilised 41 % aqueous solution of hydrogen peroxide in the presence of a proprietary chemical and a few mg/ℓ of formaldehyde and copper(16).

-the 'Degussa' version uses a 50 % to 70 % solution of hydrogen peroxide and copper as a catalyst.(16, 18, 19, 20).

Free cyanide and weak acid dissociable cyanide complexes of copper, nickel and zinc are oxidised to cyanate by hydrogen peroxide in the presence of copper. The reaction is fairly insensitive to pH, however the preferred range is 9,0 to 9,5. The base metals precipitate as hydroxides although iron cyanide is only partially precipitated. Significant thiocyanate oxidation does not occur during this process.

5.3 OZONE OXIDATION

Ozone oxidises thiocyanate, free cyanides and wad cyanides very rapidly in alkaline media.(21, 22) Thiocyanate is oxidised first followed by cyanide. Cyanate, produced by the oxidation of cyanide, is also oxidised but rather more slowly. Ozone also reacts with other impurities present in the medium and allowances must also be made for its spontaneous breakdown to oxygen(23).

The ozone consumption in cyanate oxidation is far above the theoretical value and as the cyanide ions are oxidised and cyanate ions build up in the solution, the process rate decreases and the specific ozone consumption increases. The complex cyanides of copper, iron and other metals are much less amenable to ozone decomposition. These compounds oxidise more slowly and require more ozone. Copper ions act as catalysts on the oxidation rate of both simple cyanides and complex copper and iron cyanides. Nickel ions in concentration of 10-12 mg/ℓ are also catalytically active(23).

5.4 SULPHUR DIOXIDE/AIR OXIDATION

Details of this process were developed and introduced by INCO Metals Company Ltd(24).

In this process air (or oxygen) and sulphur dioxide (as a gas or sulphite) are introduced into the cyanide solution in alkaline conditions (pH 6-11) and in the presence of copper as a catalyst. All cyanide species are destroyed except the most stable cyanide complexes of iron, cobalt and gold. Iron cyanide does precipitate as ferrocyanide complexes. Thiocyanate is oxidised to a small extent. If substantial concentrations of nickel cyanide are present, thiocyanate may be converted to cyanide(16).

5.5 FERROUS SULPHATE

The addition of ferrous sulphate to a cyanide solution will result in the precipitation of soluble cyanides as insoluble Prussian Blue. The reaction proceeds optimally at a pH between 5,5 and 6,5, at ambient temperatures (< 20 °C is preferable) and with an excess of the ferrous ion(25).

In alkaline conditions insoluble Prussian Blue decomposes rapidly, eventually yielding hydrogen cyanide. The insoluble form of Prussian Blue is unstable with respect to oxidation and pH in the aqueous phase. This instability is enhanced in the presence of light and heat which is of consequence if this process is used to inactivate cyanide in backfill operations. Decomposition of Prussian Blue in acid conditions will also yield hydrogen cyanide.

Ferrous sulphate reacts with free and weak acid dissociable cyanides to form ferrocyanide and insoluble complexes such as Prussian Blue. In acid conditions ferrocyanide will eventually release hydrocyanic acid.

5.6 ACTIVATED CARBON

Active carbon is used to treat industrial waste streams containing small quantities of potentially toxic materials, particularly if they are organic.

Some pilot plant work has been done to apply this work to the removal of cyanide from gold mine effluent streams.(26, 27) This work has indicated that copper (in granular or powder form) in mono- or divalent form dramatically improves the cyanide oxidation capabilities of active carbon.

5.7 BIODEGRADATION

Biodegradation of cyanides and biosorption of toxic heavy metals from metallurgical process effluents in order to meet environmental permit limits has been achieved using the bacteria *Pseudomonas Spp.*(28, 29, 30). The bacteria were used to form a thin film on waste water in an oxygenated environment. Free cyanide, thiocyanate, metal complexed cyanides and ammonia were removed via biological pathways through oxidation.

5.8 AVR PROCESS

In the AVR process the cyanide laden slurry is acidified, converting most cyanides to hydrogen cyanide. This compound is stripped from the slurry by air and the cyanide laden air is scrubbed with a suitable alkaline solution. This process consists of the following stages:

5.8.1 Acidification

The slurry or clarified solution is acidified using sulphuric acid. The retention time and the pH value are determined for each application. Hydrocyanic gas is evolved and is vented to the stripping system described later.

5.8.2 Volatilisation

The acidified slurry passes down a series of packed towers counter current to air. Based on published pilot plant work recoveries of better than 90 % can be achieved.[31, 32]

5.8.3 Recovery

The cyanide laden air stream passes through an absorption tower in which the cyanide is absorbed in a suitable alkali (e.g. sodium hydroxide or slaked lime slurry).

5.8.4 Re-Neutralisation

The barren slurry, now stripped of most of its cyanide, is neutralised using lime or sodium hydroxide. Base metals in solution are precipitated as hydroxides.

The AVR Process has been modified to the Cyanide Regeneration Process (CRP) and includes the use of a flocculant in the acidification step(33).

5.9 THE ION EXCHANGE PROCESS

In this process the barren cyanide solution is treated with zinc sulphate which converts all free cyanide to the zinc complex. The slurry passes counter current to the ion exchange resin in four stages. The resin is transferred from stage to stage by means of airlifts. After washing on a linear belt screen the resin is transferred to an intermediate storage tank. The barren pulp passes to waste or backfill generation. The loaded resin passes from the storage tank to elution columns where it is eluted with a solution of sodium or calcium zinc tetracyanide. This compound displaces all base and precious metal cyanides from the loaded resin which becomes loaded with zinc tetracyanide. A bleed stream is acidified to produce hydrogen cyanide (which passes to an alkaline absorption column) and a base metal sulphate solution.

Once the resin has been eluted, it is washed in the column then treated with dilute sulphuric acid and sparged with air. Hydrogen cyanide which is liberated during this process passes to the alkaline absorption column. The liquid phase (essentially zinc sulphate) returns to the complexation stage. Once the resin is regenerated, it is washed and stored for reuse.

5.10 CHELATING RESINS

The Vitrokele resin has the ability to capture metals, both total and weak acid dissociable cyanides as well as thiocyanate from both slurries and solutions(34). The process is very similar to the ion exchange process.

6. CONCLUSIONS

Based on the manner in which cyanide is used and cyanide bearing waste streams are generated, managed and disposed of, together with an understanding of cyanide chemistry and the constituents found in mine effluents, it is concluded that the potential for serious negative health or environmental impacts is minimal.(2) In all respects except ecotoxicity, mine effluents should be classified as non hazardous.

Regarding the elimination of cyanide in mine effluents, processes which recover cyanide are more economically justifiable than processes which destroy cyanide, provided that there is sufficient cyanide present to make it worth recovering. The ion exchange process offers the additional promise of gold recovery potential.

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RADIOACTIVITY

1. INTRODUCTION

The topic of radioactivity has received increasing public and media interest in the recent few years. As many South African gold mines extract ore with an economically viable concentration of uranium, the workplace, the public and the environment are exposed to potential contact with the products and discard material.

It is known that uranium is present not only in the quartz-pebble conglomerates of the Witwatersrand group, but also in other widespread mineral deposits as well as in certain coal deposits, in particular in the Springbok Flats. A comprehensive radiological survey for the entire mining industry is being undertaken by the Council for Nuclear Safety. At this stage, progress has been limited due to limited availability of personnel and analytical capability(7).

2. CHARACTERISTICS AND OCCURENCE OF RADIONUCLIDES

Concentration levels of uranium in the Witwatersrand and Far West reefs range from below 50 g/t to about 200 g/t of U_3O_8. At Beisa at the southern end of the OFS goldfields, uranium grades reach 600 g/t of ore, and at Afrikander Lease near Klerksdorp, uranium grades reach 640 g/t of ore. This contrasts with the situation in areas mined primarily for uranium such as Canada, USA and Australia, where ore grades of between 2 % and 4 % uranium are extracted. There is therefore a difference of about two orders of magnitude in the radonuclide concentrations in Canadian, USA and Australian tailings compared with those in South Africa.

Uranium decays into a series of elements that are of environmental and health importance, as seen in Table 1.

The most common decay processes are alpha (α) and beta (β) decay and gamma radiation (γ). Alpha decay is the process by which the parent nucleus transforms by emitting an alpha particle, a twice ionised helium nucleus, that carries away with it two units of nuclear charge (atomic number) and four mass units. The alpha particle, having a smaller mass than the nucleus but the same momentum, has much more energy. It does, however, have a very short range and low penetration and is unable to pass through even the outer dead layer of skin of a human being. Therefore alpha particles do not pose an external threat to the body. If they are allowed to enter the body via ingestion, inhalation, or an open wound, an alpha source can pose a health threat.

Table 1 : ^{238}Uranium Decay Series(7)

			Major radiation energies (MeV) and intensities		
Nuclide	**Historical Name**	**Half Life**	**Alpha**	**Beta**	**Gamma**
^{238}U	Uranium I	4,51 x 10^9 yr	4,15 (25 %)		
	(Uranium)		4,20 (75 %)		
^{234}Th	Uranium X	24,1 days		0,103(21 %)	0,063(3,5 %)
	(Thorium)			0,193 (79 %)	0,093 (4,0 %)
^{234}Pa	Uranium X	1,17 min		2,29 (98 %)	0,765 (0,3 %)
	(Proactinium)				1,001 (0,6 %)
^{234}U	Uranium II	2,47 x 10^6 yr	4,72 (28 %)		0,053 (0,2 %)
	(Uranium)		4,77 (72 %)		
^{230}Th	Ionium	8,0 x 10^4 yr	4,62 (24 %)		0,068 (0,6 %)
	(Thorium)		4,68 (76 %)		0,142 (0,07 %)
^{226}Ra	Radium	1062 yrs	4,60 (6 %)		0,168 (4 %)
			4,78 (95 %)		
^{222}Rn	Emanation	3,82 days	5,49 (100 %)		0,51 (0,07 %)
	(Radon)				
^{218}Po	Radium A	3,05 min	6,00 (100 %)		
	(Polonium)				
	Radium B	26,8 min		0,65(50 %)	0,295 (19 %)
	(Lead)			0,71(40 %)	0,352 (36 %)
^{218}Bi	Radium C	19,7 min		1,0(23 %)	0,609 (47 %)
	(Bismuth)			1,51(40 %)	1,12 (17 %)
^{218}Po	Radium C	0,164 sec	7,69 (100 %)		0,799 (0,014 %)
	(Polonium)				
^{210}Pb	Radium D	21yrs		0,016(85 %)	0,047 (4 %)
				0,061(15 %)	
^{210}Bi	Radium E	5,01 days		1,16 (100 %)	
^{210}Po	Radium F	138,4 days	5,305 (100 %)		0,803(0,001 %)
	(Polonium)				
^{208}Pb	Radium G	Stable			
	Lead				

Beta decay is the process of nucleus transformation by emission of a negatively charged electron (beta particle) that leaves the progeny with the same mass and one unit greater in atomic number. Generally considered to be 100 times more penetrating than alpha radiation, beta radiation can have sufficient energy to penetrate the skin and thus constitute an external radiation hazard. Internal beta emitters are definitely a hazard.

Either alpha or beta decay may produce a progeny nucleus in an excited state. As a means of this high-energy (ground) level, a photon of electromagnetic energy is typically released; a gamma ray. Gamma rays, 10 000 times more penetrating than alpha radiation, are similar to X-rays in behaviour. Gamma radiation intensity is equal to the difference between the excited- and lower-state energy levels. The alternative process for reaching a lower energy level (which competes with gamma ray emission) is internal conversion (which renders X-ray emission). The penetration of gamma rays is quite different from the penetration of alpha and beta particles because those particles can be completely absorbed by matter of a certain thickness. Gamma rays are steadily attenuated in matter, but some photons will pass through even thick shields.

People are exposed daily to radiation from sources including ^{222}Rn and its progeny, natural background, medical diagnostic practices, weapons testing, building materials, drinking water, tobacco smoke, and food. Background consists of radiation from cosmic rays, their interaction with the earth's atmosphere, and the primordial radionuclides in the earth's crust.

It is clear that the major radiation exposure to humans is from ^{222}Rn and its progeny, through inhalation. Recently, the Committee on the Biological Effects of Ionising Radiation published a report devoted exclusively to ^{222}Rn and other internally deposited alpha emitters. In general, it is rapidly becoming recognised that ^{222}Rn and its progeny are the major radiation threat to the general population. Radon in homes is widespread and is present at much higher concentrations than recognised only a few years ago.

Elevated levels of ^{222}Rn are almost exclusively confined to groundwater sources. ^{222}Rn is generally higher in the low yield, granite bedrock wells, ranging from <10 pCi/ℓ in many surface water supplies to more than 1 x 10^6 pCi/ℓ in a private groundwater supply. Extremely high concentrations are generally found only in small public groundwater supplies; non community, non-transient groundwater supplies; or in private home groundwater supplies.

In a USEPA study of public groundwater supplies, elevated levels of ^{222}Rn were found primarily in Virginia, North and South Carolina, New England, and at scattered locations throughout the Midwest and western parts of the United States(43). Only localised areas of the eastern United States showed relatively high levels of ^{222}Rn (10 000 pCi/ℓ and higher). This study, however, included analysis of public groundwater supplies serving 1,000 people or more. Cothern reported the averages for US water supplies at 780 pCi/ℓ and 240 pCi/ℓ for systems serving less than and greater than 1 000 population, respectively. Cothern cited values for the overall US-population-weighted-average ranging from 200 to 600 pCi/ℓ.

Although extremely high levels of ^{222}Rn in public water supplies are commonly perceived as being confined to a few areas of the United States, available data show that there are private drinking water supplies in at least 11 states with >50 000 pCi/ℓ and supplies in at least 8 states with >100 000 pCi/ℓ. The belief that high levels of ^{222}Rn in private or small public water supplies are confined to a few geographic areas appears to be largely a result of an extremely poor data base for the country as a whole.

If the USEPA sets an MCL for ^{222}Rn of the order of 160 pCi/ℓ, then most US groundwater supplies will not be in compliance and most well supplies will be considered high.

Radium is found primarily in groundwater supplies. Compliance data show that it is most often found in four aquifer regions : Coastal Plain, Piedmont, North Central and Colorado Plateau. Using compliance data from 50 000 public water systems, the USEPA estimated that of those in which gross alpha exceeded 15 pCi/ℓ, 500 systems exceeded the radium MCL of 5 pCi/ℓ. The highest ^{226}Ra concentration reported was approximately 200 pCi/ℓ. However, few supplies exceed 50 pCi/ℓ, and about two thirds of the supplies exceeding 5 pCi/ℓ are below 10 pCi/ℓ. The population-weighted average in public water supplies is estimated to be approximately 0,3 to 1,0 pCi/ℓ.

In contrast to concentrations in public supplies, little is known about the Ra levels in private household supplies. Concentrations as high as 200 pCi/ℓ have been reported. In addition, areas of the country not known for high Ra concentrations in public supplies have household wells with elevated Ra levels. Lucas pointed out that many private homes, farms, and housing developments on private well supplies are exposed to high Ra concentrations, making the total population exposed to high Ra greater than that identified by studies considering only public water supplies (55).

Uranium is found in both surface water and groundwater supplies, primarily in the western United States. Natural uranium includes three isotopes in the following proportions by mass: 0,006 percent ^{234}U, 0,72 percent ^{235}U, and 99,27 percent ^{238}U. Uranium activity averages are estimated by assuming that the uranium in the sample has the specific activity of natural uranium in equilibrium, approximately 0,67 pCi/µg. This is not necessarily so because of varying isotopic proportions between ^{238}U, ^{235}U, and ^{234}U that can occur. Cothern and Lappenbusch estimate the average uranium concentrations in public surface water and groundwater supplies to be 1,0 and 3,0 pCi/ℓ, respectively(56). The highest population-weighted averages occur geographically in the region from Montana to Texas and California to Kansas. Based on the results of a USEPA study of public groundwater supplies, 1,15 million people are served by drinking water supplies containing more than 5 pCi/ℓ Ra, the current MCL. Approximately 300 000 people are served by water supplies exceeding 10 pCi/ℓ U. Thorium levels were extremely low. A supply containing more than 0,1 pCi/ℓ Th would be unusual, and one with more than 1,0 pCi/ℓ Th would be extremely rare.

As a result of extensive nuclear transmutation and disintegration experiments, more than 800 new nuclides have been produced and studied. Of these, approximately 200 are considered potential drinking water contaminants. Animal and human studies have identified adverse effects to various organs and have determined that the 200 man-made radionuclides are carcinogenic.

The routes through which the man-made radionuclides may reach drinking water supplies are primarily nuclear weapons testing (fallout), nuclear power plant accidents or normal operating discharges, discharges from medical facilities, and leaching from radioactive waste facilities.

Fall out radionuclides have been distributed over the entire surface of the earth. Groundwater is highly unlikely to be vulnerable to induced radioactivity because it is inaccessible to direct fallout and because groundwater remains underground for a considerable period of time. This allows the short-lived isotopes to decay before the water is used. The long-lived radionuclides such as ^{137}Cs and ^{90}Sr are tightly bound by soil minerals following deposition and do not enter groundwater. Neither ^{90}Sr nor ^{137}Cs has been detected in groundwater as a result of nuclear weapons testing. In a study of groundwater supplies following the Chernobyl accident, those supplies deemed most vulnerable did not show an increase in radioactivity.

Considerable data are available on gross beta activity and ^{90}Sr levels in surface waters in the United States. Of the two contaminants most likely to be introduced to a surface water supply because of fallout, ^{90}Sr and ^{137}Cs, ^{90}Sr is more like to occur in natural waters because of its higher wash off rate. This is further supported by measurements taken from New York City tap water that indicated a ^{137}Cs to ^{90}Sr ratio of about 0,1, even though there is 50 percent more ^{137}Cs than ^{90}Sr in fallout and their half-lives are approximately equal.

Plutonium is quite reactive in water and when found in surface waters is generally associated with sediment. In Lake Michigan, 97 percent of the ^{239}Pu is found in the sediment, whereas the incorporation of ^{90}Sr has been measured in the Great Lakes to range from 0,5 to 1,0 pCi/ℓ.

From October 1958 to early 1964, the US Public Health Service drinking water standard of 1 000 pCi/ℓ gross beta was exceeded 40 times. Of these occurrences, 20 were cases in which the radioactivity was associated with suspended solids and would have been removed before use as a drinking water supply. Eighteen of the remaining 20 values were from the Columbia River at Pasco, Washington, where the cooling water from the Hanford Project is discharged. Here, the radioactivity was associated with the dissolved solids in the water. The remaining two values were from the Clinch River in Kingston, Tennessee, just below the Oak Ridge National Laboratory. Straubb concludes from these data that except for the possibility of higher intake of gross radioactivity and of ^{90}Sr for short periods of time, water supplies do not contribute greatly to the overall exposure of population groups - less than 20 percent of the ^{90}Sr taken into the body is ingested with water.

2.1 RADIONUCLIDES IN MINE WATER

Chilled mine service water is often used for drinking purposes instead of other water specifically provided for this purpose. For this reason, radionuclides in mine water pose a potential health problem and need to be monitored and removed as cost effectively as possible.

Underground mine waters which are in contact with uranium-bearing reefs can be contaminated to a high degree. Values as high as 8,2 Bq/ℓ and 12 Bq/ℓ for ^{226}Ra have been found in underground mine water at one West Rand mine. Keeping in mind that at one stage 26 mines were feeding 17 uranium recovery plants it can be assumed that the contamination of mine water is widespread[7].

In the OFS Goldfield, the water supply to a number of mines is augmented by surplus fissure water from the Beatrix mine over a distance of up to 40 km. Beatrix is situated next to the closed-down formerly uranium-producing Beisa mine which has now been re-opened as the Oryx gold mine; mining different ore bodies. Whether or not fissure water pumped from Beatrix to the St Helena, Unisel, Pres Brand and Pres Steyn mines can be used as water supply without treatment to remove radioactive constituents should be established(7).

Dams, lakes and streams , some of them in close proximity to old gold residue deposits to which the public has access, show *radium concentrations* close to those of natural waters (0,02 to 0,06 Bq/ℓ). Radium concentrations in water bodies with free access to the public and situated in the vicinity of mine dumps are shown in Table 2.

Table 2 : ^{226}Ra Content of Water Bodies on the East Rand with Free Access to the Public(7)

Sampling site	Bq/ℓ
Victoria lake	0,0074
Elsburg stream	0,0629
Elsburg dam	0,0185
Boksburg lake	0,0004
Geduld dam	0,0222
Cowles dam	0,0296
Nigel dam	0,1110

Contamination of surface waters can occur by direct discharge of large volumes of radioactive process water or underground mine water into surface streams which by-pass a tailings dam with its adsorption capacity for ^{226}Ra(7).

Increases in ^{226}Ra levels in surface streams have been reported. In a few cases where ^{226}Ra concentrations in effluents of up to 1,4 Bq/ℓ were measured, these fell to levels of about 0.03 Bq/ℓ within a distance of 7 to 8 km from the point of release, most probably due to adsorption of radioactive material by the bottom sediment of the stream (Table 3).

Table 3 : ^{226}Ra Content of Some Surface Streams(7)

Site			Bq/ℓ
East Rand	ERGO-Brakpan	Waste stream, leaving property	1,400
	Rietspruit	After confluence with ERGO effluent	0,222
	Rietspruit	At N103 crossing 8 km downstream	0,026
West Rand	Western Areas	Leaving mine property from North Shaft	0,190
	Western Rietspruit	About 15 km downstream of North Shaft	0,037
	Western Areas	Leaving property from South Shaft	0,222
	Leeuspruit	About 8 km downstream of South Shaft	< 0,037

Radon gas emanates from any ore body containing uranium and may be found even in non-uranium-producing mines in significant quantities to constitute a health hazard to workers. In addition, radon can be transported in underground waters and released far from the point of emanation in sufficient quantities to constitute a health hazard in mines.

Indications are that in terms of mine environmental pollutants, radon and its daughter products are the main contributors to the excessive incidence of lung cancer amongst miners. Human data are now available from several groups of underground metal-ore miners at USA, Canadian and Czechoslovakian uranium mines, Swedish and British iron mines, Swedish lead and zinc mines, and Newfoundland fluorspar mines. Although other potential carcinogens such as diesel smoke, traces of As, Ni or Fe ore are found in these mines, the lung cancer response appears to be predictable, based on Rn-progeny exposure(57). Lung cancer- hazard from inhalation exposure to Rn and Rn-progeny is apparently not from the Rn *per se* but rather from the alpha-dose delivered through lung deposition of the short-lived progeny of radon. Two alpha-emitters, ^{218}Po (RaA) and ^{214}Po (RaC) ultimately deliver the carcinogenic dose to basal cells of the bronchial epithelium, the critical tissue for induction of lung carcinoma(7).

Historically, inhalation exposure is defined in terms of the air concentration of Rn progeny in working level (WL) units. A working level is defined as U-concentration or short-lived Rn progeny totalling 1,3 x 10^5 MeV of potential alpha-energy per litre of air. A working level month (WLM) is an exposure equivalent to 1 WL for 170 h(7).

Radiological conditions during surface operations are shown in Table 4.

Table 4 : Radiological Conditions Prevailing In Various Surface Plants(7)

Type	Dose rate mrem/h	Long lived airborne alpha activity
Uranium plant	0,05 to 0,5	1 DAC*
Sulphuric acid plant	0,05 to 2,0	
Calcining facility	0,1 to 4,0	0,01 to 10 DAC
Phosphate (Baddeleyite)	0,05 to 2,0	up to 8 DAC
Monazite (Mineralands)	0,05 to 2,0	up to 20 DAC
Monazite (Mine)	0,5 to 10	up to 200 DAC
Uranium (High thorium content ore)	0,5 to 300	up to 10 DAC

DAC = Derived air-borne concentration (ore dust and uranium concentrate aerosols equal to the inhalation during a whole year or about 2 000 exposure h/a representing the annual dose.

In view of the growing concern locally and internationally with regards to radiation hazards, a comprehensive and systematic assessment of this situation in the mining industry - has been undertaken by COMRO/Miningtek although this information is not available. The Nuclear Energy Amendment Act (Act No 56 of 1988) established the Council for Nuclear Safety with regulatory powers over nuclear installations and activities involving nuclear-hazardous materials(7).

The concentration of radium in tailings is small, about 50 µg of radium/t of tailings for a 0,02 % ore. Analyses of tailings dams to assess the prevailing ^{226}Ra levels gave values of 0,6 to 3,1 Bq/g of ^{226}Ra for the Klerksdorp and OFS mining areas where the richer uranium ores associated with gold are to be found, and of 0,2 to 0,6 Bq/g on the East Rand (Chamber of Mines of South Africa, 1979)(7). Consequently, treatment of mine and tailings effluents by $BaCl_2$ to precipitate dissolved radium has not been practised in South Africa, since the radium concentrations in waste streams have been considered too low to justify such treatment(7).

On the Witwatersrand (South Africa), ore contains varying amounts of uranium and thorium and the radioactive isotopes from their decay chains. These include

- mining and processing of uranium - and thorium - bearing areas;
- other operations involving such materials including, for example monazite process-processing plants, as well as burning of pyrites in sulphuric acid plants(7).

2.2 RADIONUCLIDES IN DRINKING WATER

Because the primary function of mine service water is not drinking, specific problems related to radionuclides in drinking water will be discussed and emphasis is placed on educating the workers not to drink mine service water.

In the United States there are around 60 000 community water supplies, of which some 500 drinking water supplies (almost all groundwater supplies) exceed the 0,19 Bq/ℓ concentration limit for ^{226}Ra, and up to 650 supplies exceed the uranium concentration of 0,74 Bq/ℓ. Radon concentrations in drinking water in the USA range from a few Bq/ℓ to more than 100 000 Bq/ℓ in the North-east, with the average value being between 18,5 and 185 Bq/ℓ. In areas where the Rn concentration is high, private wells probably deliver more Rn into the household than do community water supplies. Yet, the EPA is still in the process of developing revised regulations for radioactivity in drinking water under the Safe Drinking Water Act, due to the difficulty in obtaining information from all related areas and disciplines(7).

^{222}Rn presents the most serious health threat, in drinking water, of all the radionuclides. Radon in drinking water results in internal whole-body exposure from direct consumption, that may be a significant fraction of the maximum annual critical organ dose of 1700 µSv/yr (170 mrem/yr). Radon, an inert gas rapidly enters the blood system, perfusing all the cells of the body. In the case of radon consumed in water, the radon and insoluble short-lived daughters irradiate all cells, as opposed to inhaled short-lived daughters that primarily irradiate the tracheal-bronchial epithelium(12).

Radionuclides in drinking water are believed to pose a relatively low exposure level to the general population. It has been shown, however, that radionuclides in drinking water can cause cancer and hereditary effects. Deterministic effects are not likely because the actual radiation dosages received are normally low; an exception is the case of natural uranium, which can cause a chemically toxic effect on the kidneys.

The natural radionuclides that are of interest in drinking water are found within the decay series of uranium (^{238}U), actinium (^{235}U), and thorium (^{232}Th). The ^{238}U decay series is of primary concern in terms of health effects, far exceeding the population's exposure from ingestion of isotopes in either the ^{232}Th or ^{235}U series. Briefly, the subseries of most concern are (1) ^{238}U to ^{226}Ra because of long-lived, bone seeking alpha emitters, (2) ^{222}Rn to ^{214}Po because of the internal alpha dose to the lungs, (3) ^{214}Pb to ^{214}Bi because of external and whole body dose, and (4) ^{210}Pb to ^{210}Po because of alpha decay at the bone site.

^{222}Rn and its progeny pose the largest health hazard, especially because of the dose to the lung. It is not important in drinking water through ingestion, but it contributes to the air ^{222}Rn through its release during normal water use in the home. The health risks for airborne ^{222}Rn are so significant that the USA Office of Drinking Water is considering setting the new ML for ^{222}Rn as low as 160 pCi/ℓ.

Of the radionuclides cited for potential health hazard, ^{222}Rn and its progeny are of most concern, followed by radium, uranium, and thorium and their progeny. To address the potential effect of a particular radionuclide on the population, it is necessary to determine whether it occurs in drinking water at levels sufficient to cause concern.

2.3 RADIONUCLIDES IN WATER USED FOR AGRICULTURE

The concern for radioactivity in agricultural water revolves around uranium and radium, and an increased concern for radon is being experienced. This is the case in water from uranium mines and water from natural sources(5).

Uptake of radon through irrigation on food crops is however unlikely as most radon will be lost through aeration during irrigation(5).

Concentration ratios for radioactive nuclides are needed to put in models for dose assessment. If there are farming activities in the Witwatersrand with radioactive nuclide concentration ratios larger than 1, it will contribute to the population dose(5). A specific study aimed at determining the dose contribution of radioactivity in mine effluent to man via aqueous pathways was conducted on the East Rand. After a preliminary survey, it was decided to find a locality where farming activity was as close as possible downstream from mining activities. This was motivated by rapid decline in radium concentration after mine water release. Subsequently higher levels of radioactivity were identified on the West Rand, and vegetable samples were taken from this area too. On the East Rand, the investigation focused on the Blesbokspruit area (5).

Before selecting a site for research on radionuclides in agricultural waters, the data from an airborne gamma survey by helicopter was studied. From this data it could be seen that, gamma radiation increased from east to west. In the Stilfontein area radiation was three times higher than in the Nigel - Springs area, and in the Krugersdorp - Westonaria area twice as high as in the Nigel - Springs area(5).

The increase of concentration of ^{210}Po in beef liver from cattle raised on irrigated land downstream of a mine to a level double that of control samples has also been confirmed (NUCOR, 1984)(7).

The preliminary conclusions of the research by Bain *et al*, were that low concentrations of uranium and ^{226}Ra in fields and irrigation water that approaches environmental sample concentrations, as well as the fact that the studied food pathways might have concentration ratios of between 0,1 and 0,01, the nuclides in this pathway will probably pose insignificant risk to the population of the Witwatersrand via the food pathway. Eventually, the occurence of radium and uranium in three aquatic environments affected by gold and uranium mining activities was investigated(5).

In order to evaluate the potential effects of these radionuclides on man, a dose assessment model was used to study various potential pathways of both radionuclides to man including via the soil, drinking water, irrigated vegetables, a cereal and fish. Field results show that in virtually all cases the presence and concentrations recorded for both radionuclides were at least an order of magnitude lower than concentrations found in countries such as Japan, Germany, and the USA. In all cases concentrations also fell below the maximum recommended guideline values laid down by the Council for Nuclear Safety in South Africa. The dose assessments for the scenarios chosen indicated that the annual effective dose for uranium and radium is a fraction of the allowable limit for humans.

2.4 RADIONUCLIDES IN THE AQUATIC ENVIRONMENT

The principal form in which radium is present in the tailings is $RaSO_4$, but a strong likelihood exists that the radium ion is also adsorbed on to the surface of particulate material. Laboratory leaching tests using columns of 1250 mm depth, packed with different types of tailings and onto which predetermined doses of ^{226}Ra had been adsorbed have shown that after 550 days of continuous percolation of water through the columns, radium had moved only 25 mm. Little of the original radium in the bulk of the tailings was leached(7).

The results for all test columns are consistent amongst themselves and point to an extreme low mobility of radium in the tailings impoundments studied. In situ tests on tailings dams also indicate that radium moved 1300 to 2900 times more slowly than water, which could explain the rather low radium content

(0,06 Bq/ℓ) of seepage water measured at the base of some impoundments[7].

All uranium extraction locally (in South Africa) is done by means of acid leaching. Oxidation of pyrite in the tailings results in the production of H_2SO_4 which may solublize uranium and its daughter products such as radium. Again, laboratory investigations did not show an increased leaching rate of radium with increased acidity[7].

Contamination of surface waters by ^{226}Ra has not yet been a problem in the vicinity of sand dumps and of tailings dams from gold processing only. The fact should, however, be taken into account that these deposits from gold only recovery are too dry to produce significant amounts of acidity[7]. New data collected in a recent survey, indicates that there may be a potential hazard with regard to radionuclides discharged from current and defunct mining operations into the aquatic environment[6].

Contamination of groundwater by seepage from gold tailings does not appear to be a problem, since the migration of radium in the tailings is so slow. In the case of the Buffelsfontein ground-water pollution, it has been found that the radium content of seepage into the dolomite was below 0,04 Bq/ℓ[7].

Radioactive contamination of underground water may take place due to the common practice of uranium processing plants decanting surplus water from the tailings and pumping it to unlined evaporation dams. Annual evaporation losses of 2 m/a to 4 m/a have been reported for some evaporation dams while the actual net evaporation rate should be around 1 m/a. Seepage losses into rock strata may, therefore, take place. Lining of such evaporation dams could be a remedy[7].

While ^{226}Ra is readily adsorbed on to particulate material such as sand, clay or sewage sludge and is thus removed from clear water, this is not the case with uranium, which stays in solution. Uranium can, therefore, be transported in mine service water through the mine, but also in mining effluents over wide distances[7].

At one mine, pumped-out mine water had to be returned into the adjacent dolomite compartment in order to prevent ground subsidence and the formation of sinkholes. In order to increase the life span of the mine and to reduce the high cost of pumping, this mine obtained permission to dewater the compartment and to discharge the effluent via a public stream. The average uranium level of the dolomite water still entering the mine from this compartment after three years of dewatering is 67 µg/ℓ, which is above the presently accepted standard for the public of 44 µg/ℓ uranium. Taking this situation into account, the mine is conducting a programme to measure radioactivity underground to ascertain whether any danger areas exist[7].

A preliminary survey of uranium and radium concentrations in different fauna and flora species was done on the East Rand. In only a few samples were the ^{226}Ra concentrations above the detection limit. All of the uranium values were lower than the maximum uranium ore concentrations of 200 µg/g found in this area.

Table 5 shows uranium and ^{226}Ra uptake in two wetland reed plant species (*Arundo donax* - exotic, and *Typha capensis* - indigenous) as well as the sharptooth catfish (*Clarias gariepinus* - indigenous), as summarised from Bain *et al*(5). While uranium may not always be detected in high concentrations in water and settled uranium in sediments may be eroded away, biota may serve as biological indicators of radioactivity in aquatic ecosystems. Soluble Uranium in water has a toxicity and LC_{50} concentrations for organisms are generally low. Four Australian tropical fish species showed 96 hour LC_{50}s for uranium ranging from 0,73 - 3,46 mg/ℓ (2). Radionuclide levels were measured in tissues, of beavers in the vicinity of uranium mining activities in North America, and bone concentrations of ^{226}Ra are used as an important indicator of contamination. Contamination was regionally limited to individuals close to mining activities(15). These studies of affected organisms serve as an indication of the aquatic impact of radionuclides in comparison to other pollution problems.

Two species that may serve as good biological indicators in South Africa are, *Typha capensis* and *Clarias gariepinus*, because they have a wide spread distribution and are quite common.

Table 5 : Uranium and ^{226}Ra Uptake in Selected Species

SPECIMEN	LOCALITY	^{226}Ra (Bq/kg)	U (µg/g)
Arundo donax	Elsburgspruit	<41	30
Arundo donax	Elsburgspruit	<100	1,68
Arundo donax	Blesbokspruit	<162	37,7
Arundo donax	Natalspruit(Vpt)	<87	14,2
Arundo donax	Germiston Lake	195 ± 15	18,1
Typha capensis	Elsburgspruit	<33	31,1
Typha capensis	Elsburgspruit	<41	5,98
Typha capensis	Blesbokspruit	153 ± 31	111
Typha capensis	Natalspruit(Vpt)	<76	15,3
Typha capensis	Germiston Lake	97 ± 13	12,5
Clarias gariepinus	Blesbokspruit	<36	4,99
Clarias gariepinus	Natalspruit(Vpt)	<63	0,6
Clarias gariepinus	Germiston Lake	<65	2,42

Mass balances of ^{226}Ra and ^{238}U have been used to assess the pollutant discharge from uranium mining and milling and to identify the main physical and chemical processes involved in radionuclide transport. The studies show that seasonal load variations are influenced by mobilisation of sediments during wet periods(11). This should be kept in mind when sampling river water and sediment.

The ^{226}Ra concentration in natural soils varied between 0,004 and 0,126 Bq/g in this study(5). A sediment of eroded mine slime sampled was almost an order of magnitude higher than 0,126 Bq/g.

Water samples taken from the Blesbokspruit from the point where water was being abstracted for vegetable irrigation, had uranium values below the limit for uranium drinking water, and the radium values were just above the limit of 0,19 Bq/ℓ(7).

3. STANDARDS FOR RADIOACTIVITY

Because of naturally high radioactivity levels in the USA, development of standards has been given much attention, and because standards are few and not well developed in South Africa, many authors base discussions on USA standards.

In America the National Interim Primary Drinking Water Regulations (NIPDWR) for radionuclides have been in effect since July 1976. They were set at the following levels:

- 5 pCi/ℓ for ^{226}Ra and ^{228}Ra combined,
- 15 pCi/ℓ for gross alpha particle activity, excluding ^{222}Rn and U, and
- a total dose equivalent of 4 mrem/year for man-made radio-activity.

The MCL of 5 pCi/ℓ for radium was based on a lifetime (70 years) excess cancer risk rate of 10 at a calculated dose of 150 mrem/year to the bone. Gross alpha, thought to be a more practical and economical screening test for compliance with the radium MCL, was set at 15 pCi/ℓ based on the following rationale: If the radium concentration is 5 pCi/ℓ and the rest of the gross alpha is caused by the next most radiotoxic alpha subsidies, starting with ^{210}Pb, the total dose to the bone would be less than that from 6 pCi/ℓ ^{226}Ra(18).

Ingestion of any combination of man-made radionuclides from a surface water supply must be less than that required to cause a total body dose of 4 mrem/year. This relates to a fatal cancer risk of approximately 10/year(18).

The interim regulations require quarterly sampling of public water supplies throughout one year every four years. If the gross alpha is greater than 5 pCi/ℓ then the water supply is to be analysed for ^{226}Ra. If ^{226}Ra is greater than 3 pCi/ℓ, the sample is to be analysed for ^{228}Ra. Monitoring for gross beta particle and photon activity is required of all surface water systems serving more than 100 000 people. Strontium (because of its toxicity), tritium (because it is not included in the gross beta count), and gross beta are measured. If the gross beta is greater than 50 pCi/ℓ, then the sample is re-analysed to determine the presence of other radionuclides. The combination of all man-made radionuclides found in the water sample must not cause a dose greater than 4 mrem/year. The Federal Radiation Council recommends a maximum dose of 170 mrem/year for the general population(18).

The 1986 Safe Drinking Water Act (SDWA) Amendments authorised the National Primary Drinking Water Regulations and require that the USEPA set such regulations on 83 contaminants, including radionuclides. The radionuclides under consideration were ^{226}Ra, ^{228}Ra, natural uranium (^{231}U + ^{235}U + ^{238}U), and ^{222}Rn. In addition, the USEPA considered limits for gross alpha particle and gross beta and photon activities. The USEPA is required to propose MCLs and MCL goals (MCLGs) simultaneously. The MCLGs, non-enforceable health goals, are to be set at levels at which "no known or anticipated adverse effects on the health of persons occur and which allow an adequate margin of safety." For the radionuclides, it is expected that the MCLGs will be zero. The MCLs are to be set as near as possible to the MCLGs, taking into consideration cost, practicality, and general availability of treatment technology. All public water supplies must comply with these regulations(18).

It was reported that the USEPA was considering an MCL for ^{222}Rn of 160 pCi/ℓ. A review of other recommended levels for drinking water radionuclides indicates that the suggested levels by the Health Physics Society are less conservative than the current levels the USEPA is considering. This is largely a result of differences in the acceptable risk levels rather than a disagreement on the estimated health effects(18).

The USEPA prescribes a screening level of 50 pCi/ℓ for beta and photon emitters (fission products) and ^{40}K, ^{241}Am, and ^{239}Pu. Only 4 of the 200 radionuclides yield a risk of 4 mrem/year at concentrations less than 100 pCi/ℓ. They are : 40 pCi/ℓ ^{239}Pu, 60 pCi/ℓ ^{115}In, 80 pCi/ℓ ^{134}Cs, and 50 pCi/ℓ ^{90}Sr(18).

Available data on the man-made radionuclides in drinking water supplies show no occurrence above the recommended screening level of 50 pCi/ℓ. Considering the relatively low estimated health risk values of the USEPA and the fact that ^{230}Pu is the only radionuclide with a concentration of concern less than 50 pCi/ℓ, man-made radionuclides are of little consequence in drinking water. This is certainly true in comparison with natural radionuclides(18).

Indications in the late 1950s were that ^{226}Ra had a radio toxicity level of about ten times that of the next most toxic nuclide in the uranium decay chain. Hence the concentration of ^{226}Ra came to be used as a common measure of radioactivity in mine water and effluent. However, later reviews on the radiotoxicity hazard at occupational exposure levels - which are about 200 times the public permissible exposure level - have shown that uranium, thorium and polonium nuclides are of higher toxicity. ^{226}Ra is therefore now considered to be the second most toxic nuclide (7).

Due to the only recently acquired knowledge of radio toxicity hazards, limited measurements of nuclide radio toxicity levels other than ^{226}Ra have been carried out. In general, analytical methods to determine radioactivity hazards are complicated, long-winded and costly(7).

Results of recent surveys have indicated levels of radium and uranium in many effluents leaving mining property which are in excess of the following accepted standards for the public, namely:

^{226}Ra: 0,19 Bq/ℓ

Uranium: 44 µg/ℓ

The measured radium and uranium levels in effluents for some mines are given in Table 6 (Council for Nuclear Safety, 1988/89).

Uranium is measured colorimetrically and is expressed in µg/ℓ. This method is more convenient and preferred because establishing the radiation emission from uranium is laborious and long-winded. Unfortunately only grab samples had been taken and the volume of the waste flows was not recorded. The data recorded in Table 6, therefore, indicate only the presence but not the extent of radioactivity.

Table 6 : Radium And Uranium Concentrations In The Effluents From Some Mines (Metcalf, 1990)$^{(7)}$

Site		^{226} Ra (Bq/ℓ)	Uranium (µg/ℓ)
East Rand	A1	0.67	618
	A2	0.66	151
	B	0.12	47
	C1	0.18	336
	C2	0.60	539
	D	0.21	262
West Rand	E	0.22	16
	F1	0.03	15
	F2	0.05	118
Far West Rand	G	0.31	305
	H	0.48	204
	I	0.03	402
Orkney/Klerksdorp	J	1.23	45

There are at present, no limits set for radioactivity levels within General Standards for the disposal of effluents into public streams nor in the SABS specification for drinking water$^{(7)}$. The SABS recommends compliance with ICRP limits. Proposed drinking water criteria for uranium are $^{(19)}$:

- Maximum limit for no risk : 1 mg/ℓ (24,79 Bq/ℓ)
- Maximum permissible limit : 4 mg/ℓ (99,16 Bq/ℓ)
- Crisis limit : 8 mg/ℓ (198,32 Bq/ℓ)

Recent (1987) recommendations by the International Commission on Radiological Protection (ICRP), reaffirmed the limit on annual dose equivalent to whole body for workers of 50 mSv introduced in 1956. The new recommendation incorporated three important developments.

Firstly, the introduction of effective dose equivalent, which was intended to be proportional to stochastic risk for either uniform or non uniform irradiations of the whole body. Doses from external and internal exposure could be combined on a common risk basis. Therefore, the annual limit of 50 mSv applies to the sum of effective dose equivalents from external and internal exposure(10).

Secondly, quantitative risk factors, can be used to derive dose limits from an assumed limit on acceptable risk. For a risk factor of 10^{-2} Sv^{-1} the limit on annual dose equivalent of 50 mSv corresponds to an annual mortality risk of nearly 10^{-3} (10).

Thirdly, the ICRP no longer assumed that the genetic hazard was the principal concern in setting dose limits for workers. A limit on accumulative dose was not included. Maximum permissible concentrations and maximum permissible body burdens in air and water were first published in the 1950's, secondary limits are based on dose equivalent to the whole body or the critical organ (critical organ is the organ receiving the highest dose). Secondary limits are expressed as annual limits on intake (ALI's) and derived air concentrations (DAC's)(10).

For the purpose of some of the most recent studies in South Africa on mine water, available EPA (USA) data, unpublished IAEA (International Atomic Energy Agency) and the ICRP (1994) were used to determine allowable limits (see table 7))(6).

Table 7 : Water Ingestion At 2 Litre Per Day Uranium And Thorium Decay Series: Long-Lived Alpha And Beta Producing Nuclides. Committed Dose Equivalent Per Unit Intake Via Ingestion (Sv/Bq).

Nuclide	Latest Unpublished IAEA	Data from Health Handbook 1992	ICRP 68 (1994) for Workers	Total Alpha short-lived induced
	Sv/Ba	Sv/Bq	Sv/Bq	Ba/ℓ
U-238	7.6E-09	6.88E-08	4.4E-08	1
U-235	8,3E-09	7,19E-08	4,6E-08	1
U-234	8.3E-09	7,66E-08	4,9E-08	1
Pa-231	7,1E-07	2.86E-06	7.1E-07	1
Th-232	1,2E-06	7.38E-07	2,3E-07	1
Th-230	2.2E-07	1,48E-07	2,1E-07	1
Th-228	1,4E-07	1,07E-07	7,0E-08	5
Ac-227	1,1E-06	3,80E-06	1,1E-06	5
Ra-228	2.9E-07	3,88E-07	6.7E-07	0
Ra-226	3,3E-07	3,58E-07	2,8E-07	4
Po-210	1.2E-06	5,14E-07	2.4E-07	1
Pb-210	8.3E-07	1.45E-06	6,8E-07	0

* Data may vary due to solubility (blood uptake) of various chemical compounds of the nuclides concerned.

4. REMOVING RADIONUCLIDES FROM WATER

Lime softening, ion exchange and reverse osmosis have been demonstrated as removing from 93 % to 97 % Ra from drinking water. For the removal of uranium, studies on anion exchange, lime softening and reverse osmosis have been undertaken. The question arises as to how radioactive waste concentrated in the brine originating from ion exchange, in the lime softening sludge, or as reject stream from reverse osmosis or any other process should be disposed of(7).

4.1 CHEMICAL

Barium chloride can be used in effluents with a high sulphate concentration, to form a precipitate barium sulphate - radium sulphate. The precipitate is very finely divided and slow settling, so a flocculant is added. Aluminium sulphate may act as both coagulant and flocculant. If water approaches freezing point, lime may be added to restore settling rate(17).

Radium may be removed by many conventional processes. The most efficient of these are anion exchange and lime-soda softening, capable of 85 - 95 percent radium removal(18).

Ion exchange is practical and commonly available for smaller systems, even at the POE (*point of entry*) scale. It requires relatively little space, is simple to operate, and is reliable for radium removal. For cases in which exchange with Na^+ is undesirable, KCl can be used as a regenerate. In small and medium-size water plants, H^+ may be a more economical and successful regenerant(18).

Lime softening is more economical for larger water supply systems. It can, however, require complicated equipment and careful operation because the pH must be maintained at 10 or higher to optimise radium removal(18).

Other removal methods, based on the ion exchange principle, have been *investigated, but treatment techniques designed specifically for radium* removal are not in general use. In laboratory conditions, radium adsorption onto iron floc in a sand filter has achieved 85 - 90 % radium removal(18).

One promising medium for removal is the radium selective complexer (RSC), which has an extremely high capacity for radium. It can reduce or even eliminate the need for regeneration on site, which, in turn, would eliminate the concern associated with the waste stream concentration. This simple process could be applied to small or large water supply systems without increasing the corrosion potential of softened water. The practice of removing large amounts of radium without regeneration may create a potential health hazard resulting from the RSC bed becoming a ^{222}Rn generator and, of course, the RSC eventually will have to be disposed of properly(18).

4.2 PHYSICAL

After chemical treatment, physical treatment may be used, such as a multimedia sand filter system. The upper layer is anthracite coal, the middle layer is silica sand and the bottom layer is high-density garnet sand(17).

This system, in one study, removed 98,4 % of the ^{226}Ra, coming from a tailings pond. Of this treatment 89,2 % was removed by the chemical process, 7,7 % in the settling pond and ±1 % by the sand filters. Of the total ^{226}Ra received by the sand filter however, 30 % was removed. Although the efficiency of the secondary settling and filtering is low, it would be difficult to reach set standards without them(17).

Radon removals of more than 95 percent can be achieved by either of two basic processes: adsorption or aeration. The granular activated carbon (GAC) adsorption-decay process is limited to small water supplies (less than 10 000 gpd) because of the relatively long empty bed contact time (EBCT) required. Its simple operation and small space requirement make it especially practical for point-of-entry (POE) applications. However, the associated elevated gamma exposure rate caused by the steady-state build-up of ^{214}Pb and ^{214}Bi, and the long-term accumulation of ^{210}Pb and ^{210}Po are considerations that make the future use of GAC uncertain. Because of the small amount of mass involved with ^{222}Rn and its short-lived progeny at steady state, the life of a GAC bed with respect to ^{222}Rn removal is very long. This process is somewhat unique in that it will undoubtedly be other water quality contaminants that will ultimately limit the steady-state removal of ^{222}Rn. Lowry and co-workers have reported in detail on the current information available from research on this process(18).

Aeration has been used for more than 20 years to effectively remove ^{222}Rn from a large public water supply in England. A packed tower designed for a small (30 gpm) industrial potable water supply achieves greater than 95 percent removal. Aeration is much more cost effective than GAC at flows greater than a few thousand gallons per day. The Henry's constant for ^{222}Rn is approximately 2 200 atm at 20 °C, making it an easily stripped gas(18).

Various treatment systems are discussed by a number of authors, and conventional settling, filtration, reverse osmosis, ion exchange, membrane microfiltration, liming and coagulation techniques are used to remove nuclides from water. Literature reviews on neutralisation, flocculation and settling, filtration and desalination discuss these methods in detail, and will therefore not be discussed in this review.

4.3 BIOLOGICAL

Scientists have discovered various micro-organisms that use uranium the way most organisms use oxygen - to metabolise their organic nutrients. Specific bacteria - one isolated from sediments in the Potomac River near Washington

D.C., and the other from butter - may be the key to cleaning up uranium from contaminated water (1). Called GS-15, the river based bacterium converts highly soluble uranium - which is difficult to isolate in waste water - into an insoluble form that can be filtered out. In addition, GS-15 tolerates uranium concentrations as high as 2000 mg/ℓ, and might be used to treat contaminated ground water(1).

Research has shown unequivocally that bacteria can directly reduce soluble uranium(VI) to insoluble uranium (IV). Thermodynamic calculations performed indicate that bacteria that normally reduce iron (III) to iron(II) can potentially get twice as much energy by reducing uranium (VI) to uranium (IV), and the reduction is more effective than indirect reduction by hydrogen sulphide that is released by microbial metabolism(3).

5. DISPOSAL

The question of management of radioactive waste is one being given close attention by the Chamber of Mines. There is a permanent forum - that has a number of subcommittees, one of which deals with radioactive waste. There are several strategies for rendering a commodity safe, and guidance is being taken from international practice. Codes of practice are being generated. The committee bears in mind several principals: ALARA (as low as reasonably achievable), national priorities, the importance of keeping the public informed (before they are misinformed), and responsibility of the State, which must take over responsibility eventually as it will outlive any company(8).

Discarding radioactive material so it does not pose a threat to water resources is a well discussed topic that recently made news headlines. The disposal site of Koeberg power station (Vaalputs), is an example of an appropriate site, but areas with granite protecting aquifers, very low rainfall and low population densities are in the western side of the country, while most mining activity is in the eastern part of the country.

Vaalputs was selected, evaluated and designed in accordance with internationally acceptable norms for the disposal of low- and intermediate-level radioactive waste generated from the Koeberg Nuclear Power Station. The main criterion was to demonstrate that the disposal of radioactive waste will not cause harmful effects to man or beast. This implies that over a period of 300 years the radioactivity will be safely contained within disposal trenches(16).

Legislation or public interest may eventually make discarding radioactive waste in these isolated areas a consideration. Investigations into permanent underground repositories, using old mines have been investigated in Germany(13), this may be a future option in the arid areas in South Africa as the presence of groundwater is a major concern in that it provides the primary transport mechanism for long-term radioactive releases. In addition the spatial extent of pathways and their retention capacity will affect both individual and effective dose limits for all but the longest-lived nuclides. If movement through a pathway is slow and lengthy, the pathway will increase the degree of decay of the waste(15).

Gold slimes dams may contain various radionuclides. Radio active waste is a material with an activity above an agreed level and that has no foreseeable use. In the activity of mining, wastes arise in slimes, rock dumps, sludges, soils and various products of uranium plants including hardware and buildings themselves. Licensing by the Council for Nuclear Safety (CNS) for the purpose of protecting public and mine workers against nuclear hazard, has been initiated. Licensing includes laying down of standards, assessment of risk, design of controls and implementation. What should be included is not only protection of the environment but also neighbouring territories. Levels of 74 Bq/g or 3 700 Bq total activity used to be considered those at which attention is required, but a newer standard is 1 000 Bq/g ^{226}Ra or 1 000 Bq total activity. For ^{226}Ra in soil, 0,04 Bq/g is considered safe, 0,2 - 0,4 needs radon free design and typical tailings dam material is 0,6 - 4 Bq/g (waste rock 1,0 Bq/g. The CNS considers material with an activity of < 100 Bq/g as safe to dispose of in slimes, material 100 - 1 000 Bq/g could be diluted; and material > 1 000 Bq/g should be treated as radioactive waste(9).

The 40 larger active South African gold and uranium mines annually dispose of approximately 120 000 000 tons of ore and 30 000 000 tons of waste rock. The 36 smaller mines in the Eastern Transvaal process roughly 1 100 000 tonnes of ore per annum, while 90 000 000 tons p.a. of sand dumps and slimes dams are reprocessed for gold uranium and pyrite recovery(9).

High values for radioactivity were reported from one, now dismantled sulphuric acid plant at AECI, Modderfontein, apparently originating from the roasting of uraniferous pyrite. Burial of the contaminated structural material (steel, bricks, ceramic Raschig rings, rubber lining) has been accepted as the solution. There are at least ten sulphuric acid plants, which are confronted with the problem of radioactive contamination from the roasting of uraniferous pyrite(7).

The storage of $18,5 \times 10^{14}$ Bq in the 65 km^2 of slimes dams and the 15 km^2 of sand dumps on the Witwatersrand might constitute a long-term health hazard. Tailings containing uranium will constantly emit gaseous radon ^{222}Ra, which decomposes into radioactive solid short-lived decay products ^{218}Po, ^{214}Pb, ^{214}Bi, ^{214}Po, ^{210}Po (Table 4). It is claimed that inhalation of these radon decay products, attached to airborne particulates, is by far the largest contributor to total radiation exposure (90 % and over). Because radon gas has a short half-life of 3,8 days, much of it decays before it reaches the surface. Therefore, a reduction in radon gas release can be achieved by retarding the rate of diffusion with the result that much of the radon decays before reaching the atmosphere(7).

Measures stipulated in the USA for remedial action at inactive uranium mill tailings include:

- the covering of existing surface with a thick layer of earth to limit radon releases to below 0,74 Bq/m^2.s; or

- deep disposal (e.g. as backfills)[7].

Field tests have demonstrated that both thick earthern covers and thinner asphaltic covers can readily reduce radon emissions by 90 % to 99 %. This aspect has been investigated by the Atomic Energy Corporation[7].

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BOILER/COOLING WATER TREATMENT

1. INTRODUCTION

The South African gold mining industry has, since 1982 reduced its use of water purchased from authorities by about 30 %. This reduction has been achieved by increasing mine service water recycling and reducing domestic uses(1).

The use of mine service water without any form of pretreatment or in-service treatment can lead to various problems. These include corrosion, scaling, fouling and microbiological activity. These mechanisms will all seriously reduce heat transfer efficiency and in extreme cases, result in equipment failure. These problems can be identified individually, but are often inter-related; microbial activity can lead to the destruction of the treatment chemical which could result in corrosion or scaling; dirt ingress into the system if uncontrolled can deposit on heat exchange surfaces leading to corrosion and bacterial activity(2).

Untreated cooling tower water contains suspended solids which become concentrated and cause fouling and microbial growth. Cooling water treatment technology has advanced to a stage where most of the problems discussed can be overcome. Practices include solid/liquid separation (clarification), desalination, softening, disinfection, corrosion, scaling and fouling prevention(3).

Ultra pure water is produced for boiler use by treating feed water to reduce the concentration of impurities such as organic and inorganic species, suspended particles, bacteria, silica and dissolved gases. Treatment methods include the use of membrane based filtration and ultrafiltration, deionization by reverse osmosis, ion exchange, continuous deionization and electro-dialysis(4).

2. COOLING WATER SYSTEMS

The following types of cooling water systems are used industrially:

- open recirculation system.
- closed recirculation system.
- once-through system.

A closed recirculation system is used to transport heat from a heat source to a radiator which is normally air cooled. The mines use these systems for internal combustion engines and the closed cooling loops of some air compressors. Chemicals are added to these systems to protect against corrosion(3).

Once-through systems require large volumes of water and are used where such supplies are available e.g. the sea or large rivers or dams. The temperature of the water is raised slightly in the process, which can result in a heat pollution problem. Chemicals are not used for water treatment and corrosion protection is achieved by correct materials of construction(3).

The open recirculation system is used most widely in the gold mining industry. The circulating water transports heat from the heat exchange equipment to a cooling tower. The cooling is achieved by the evaporation of a portion of the circulating water. The quantity of water lost by evaporation is approximately 2 % of the rate of circulation per 10°C cooling obtained in the tower(3) .Water is also lost through windage which is a loss of fine droplets of water entrained by air. Whereas evaporation loss causes an increase in the concentration of dissolved solids in the circulating water, windage tends to limit the degree of concentration. As the dissolved solids concentrate by evaporation the solubility of some of the salts in the water is exceeded and a scale is deposited on the heat exchange surface. The term used to indicate the degree of concentration of the circulating water as compared with the make-up water is cycles of concentration(2).

There are a variety of approaches for reducing the increased total dissolved solids (TDS) level which results from recycling, and controlling the level of salts and other variables within acceptable ranges to ensure satisfactory cooling system operation.

2.1 SCALING

The factors which influence scaling are pH, alkalinity, calcium, magnesium, sulphate, phosphate, temperature, dissolved solids and silica. Hence typical scales found in cooling systems can be controlled by (2) :

- reducing the alkalinity of the water by dosing acid.
- adjusting the design and mechanical operation of the system to ensure a minimum water flow velocity of 1 m/s.
- using non-adherent material in heat exchangers to reduce the number of nucleation sites.
- air rumbling.
- installing larger heat exchangers with more surface area.
- using chemical inhibitors which reduce the growth of crystalline scale.

Inhibitors such as polyphosphates, phosphates and polycarboxylic acids are normally used to prevent scale formation(2). They are highly active and are strongly attracted to surfaces, this provides competition between the inhibitor and the crystallising substances for active growth sites. If the inhibitor is successful, crystal growth or scaling ceases.

Very small amounts of inhibitor can prevent scale formation even if the scaling salts are present in high concentrations. If insufficient inhibitor is present to render the growth sites inactive, the presence of the inhibitor distorts crystal growth and adhesion so that the formation of hard scale is prevented.

When scale inhibitors are added to the circulating water the water can be allowed to concentrate further without resulting in scaling. If calcium carbonate, calcium sulphate, calcium silicate or magnesium silicate become over saturated in the circulating water however, economic dosages of scale inhibitor will not prevent scale formation(3). The oversaturation of dissolved salts is prevented by limiting the cycles of concentration of the circulating water to within the effective range of the scale inhibitor used. This may be achieved by bleeding some of the circulating water from the system. An alternative method is side stream softening of the circulating water using lime-soda softening, reverse osmosis, ion exchange or electrodialysis. In these processes, a side stream of cooling water is treated to remove sufficient materials so that scaling and corrosion can be controlled to acceptable levels. With the exception of the lime-soda process, these technologies are expensive and produce brines that must be disposed of by solar or mechanical drying methods(5).

The lime-soda softening process removes calcium, magnesium and silica in the form of a sludge discharged from the softener. This process is most suitable for makeup waters that are not high in chlorides or dissolved solids concentrations(5). Lime (calcium hydroxide) is added to the system to raise the pH to about 11. At this pH, calcium carbonate and magnesium hydroxide are precipitated. Ions such as sodium, potassium, sulphate and chloride are not removed in the softening process.

2.2 FOULING

Although the terms fouling and scaling are often used synonymously, foulants are softer non-crystalline deposits originating from suspended material in the circulating water. These materials include silica, iron oxide, organic contaminants and microbiological slime(2). Fouling may also result from overdosing of scale and corrosion inhibitors which then stick together and settle out of the water.

Fouling in cooling water systems can be controlled by increasing the rate of blowdown, by the installation of side stream softening, by the use of the air rumbling procedure (the use of high pressure air to increase the turbulence of a water stream for mechanical descaling), or the use of chemical fouling inhibitors. Increasing the blowdown rate means increasing the make-up water rate, which can be expensive. Side stream softening by means of the hot lime-soda process reduces the suspended magnesium in the circulating water by precipitating magnesium hydroxides which aid in silica removal(6). Iron is precipitated by oxidising it to yield insoluble ferric hydroxide. The oxidation is normally carried out with oxygen/air, chlorine or potassium permanganate(6).

There are essentially two types of fouling inhibitor; charge reinforcement dispersants and wetting agents(2). The dispersants keep foulants in suspension, preventing them from settling onto the metal surfaces.

Microbiological fouling in cooling water systems are potentially the most dangerous, the slime-like deposits occur throughout the system. The slime deposits act as a filter to remove suspended solids from the water which leads to other forms of fouling. Cooling tower temperatures are well within the range in which these organisms survive and thus provide an ideal environment. There is also sufficient nutrient matter in the form of organic matter, nitrogen, phosphorus, ammonia, sewage and oxygen dissolved in the water(2).

In addition to the formation of deposits these organisms can be responsible for extreme forms of corrosion under these deposits by anaerobic bacteria. Chemicals available for the control of microbiological growth in cooling systems are the following(2) :

- oxidising biocides
- non-oxidising biocides
- biodispersants

Chlorine is one of the most widely used oxidising biocides. Non-oxidising biocides can be toxic and are not suitable for potable water. Biodispersants are used as aids to oxidising agents, they loosen deposits, thereby making destruction easier and more effective(2).

2.3 CORROSION

Corrosion is the end result of scaling, fouling and microbiological activity. In a corrosion cell the corrosion process is made up of four components(2) :

- the pure metal donates electrons at the anode.
- the cathode accepts electrons from the anode.
- the oxygen and the water combine with the electrons to form hydro-oxides.
- the cooling water containing dissolved minerals provides the link to complete the corrosion circuit.

The main factors contributing to corrosion are the following(2):

- water pH; if the pH is too low the water will be aggressive, if the pH is too high scaling will occur(7).
- the attachment of dissimilar metals.
- system deposits.
- dissolved gases; the presence of ammonia with oxygen increases the rate of corrosion of copper and its alloys.
- water temperatures; corrosion proceeds more quickly in hot water.
- dissolved solids; an increase results in a higher conductivity of the water.

- microbiological matter; increases corrosion by depositing on surfaces and generates highly corrosive by-product wastes such as hydrogen sulphide(8).

Corrosion and scaling in open recirculating systems can be effectively controlled by chemical means(9,10) :

- chromate programs.
- alkaline zinc programs.
- stabilised phosphate programs.
- all organic programs.
- synthetic anionic polymers and organophosphorous compound programs.

The type of program chosen will depend on the water chemistry as well as environmental considerations.

Chromates would not generally be considered in the mining environment, especially if the water is discharged underground, due to environmental problems associated with effluent containing Chromates.

The alkaline zinc program operates in alkaline conditions in open recirculating systems. These programs are blends containing zinc, scale inhibitors and general dispersants which act to inhibit scale, corrosion and deposit formation. This program requires precise control and is sensitive to pH upsets. The stabilised phosphate program is a blend of polyphosphate which minimises corrosion of mild steel, an inhibitor to prevent corrosion of copper and copper alloys and an inhibitor to minimise fouling. This program is also very pH sensitive and is therefore seldom recommended for mine service water which is susceptible to wide fluctuations in pH and alkalinity.

All organic programs are non-metallic multifunctional treatments which prevent corrosion and scaling. The all organic chemical treatment is able to operate at pH, alkalinity and hardness levels beyond those normally tolerable in antiscalant programs, and therefore provides an excellent choice of treatment for systems using mine service water. If persistent low pH and alkalinity levels are experienced caution should however be exercised in the use of this program.

Synthetic anionic polymers and organophosphorous compound programs consist of anionic polymers which are strong dispersants assisting in the prevention of scale and fouling deposits and organophosphorous compounds which are effective in the stabilisation of scale forming species. Due to the fact that these programs operate at alkaline pH's, the corrosion potential is lowered but the program itself offers no corrosion inhibition. Low pH and alkalinity levels of the circulating water will cause corrosion to occur at increased rates when compared to other programs.

In order to prevent system deposits from dirt which will cause anode sites resulting in pitting corrosion, various techniques such as settling, filtering and chemical treatment are used to clean or purify water. From a chemical point of view clarification is the oldest and most widely used technique. It is a process applied to water for the removal of suspended solids, finer solids and colloidal material[2]. This process incorporates coagulation, flocculation and sedimentation.

2.4 BIOLOGICAL CONTROL

Different types of habitat within cooling systems encourage the growth of a variety of organisms :

- the daylight illumination of open cooling towers allows algae to develop.
- the anaerobic conditions which occur in dead legs and slime layers favours the growth of sulphate reducing bacteria which result in microbially induced corrosion.

Cooling water systems harbour two distinct microbial populations:[11]

- planktonic (free-floating) micro-organisms present in the circulating bulk fluid.
- sessile (surface attached) micro-organisms growing in biofilms.

These two micro-organisms differ very significantly in their sensitivity to biocides. In treating a cooling system, a biocide should be chosen for its ability to specifically control or remove sessile micro-organisms and biofilms as these are responsible for most biofouling problems.

Biofilm development is affected by the following cooling water system parameters[11] :

- ambient temperature affects biofilm thickness.
- water flow rate; an increase in flow rate results in an increase in biofilm thickness unless shear forces remove attached biomass.
- nutrient availability as low as 1 ppm can lead to biofilm development.
- surface material/ roughness; a rough surface biofouls more quickly.
- system temperature; a thicker biofilm forms at more favourable temperatures.
- pH; neutral is optimal.
- concentration of inorganic particles; these provide additional attachment sites.
- bleed rate; an increase reduces nutrient concentration.
- biocide type and quantity; effectiveness of the treatment will affect the biofilm.

The major groups of microbes found in cooling systems are green algae, diatoms, blue-green algae, fungi, protozoa and heterotrophic bacteria.[12] The algae and diatoms use carbon dioxide as their carbon source, the fungi, protozoa and heterotrophic bacteria use organic material as their carbon source. Organisms that may exist under deposits in cooling systems, in poorly oxidised closed loops and cooling systems with high concentrations of ammonia are sulphate reducing bacteria and fermentative bacteria, denitrifying bacteria and nitrifying bacteria respectively.

Microbiological fouling is controlled using chemical treatment which incorporates both biocides and biodispersants. Chlorine, an oxidising biocide, is commonly used for general bacteriological control(9). Chlorine treatment is supplemented with non-oxidising biocides which contain dispersants since it has no penetrating or dispersant properties. The elevated pH conditions under which most recirculating cooling systems operate will adversely affect the chlorine's performance.

Other oxidising biocides include ozone, bromine, hydrogen peroxide, permanganate, chloramines, chlorine dioxide and certain hydantoins that donate hypobromous and hypochlorous acid(12). Non-oxidising biocides include a variety of proprietary organic molecules e.g. carbonates, thiosulfanate, propyl diamine. The following factors must be considered when using non-oxidising biocides:

- required contact time.
- minimum concentration necessary to achieve kill.
- compatibility with scale/ corrosion inhibitors.
- half life at the pH and temperatures of the cooling system.

Additional information is given in the literature review on disinfection.

3. BOILER FEED WATER TREATMENT

A boiler water system consists of make-up water, which is purified raw water, and return water from steam condensate. Boiler feed water is treated to eliminate materials that cause scale, corrosion and caustic embrittlement. The maximum permissible contaminant level in boiler water is related to steam pressure; high pressure boilers require a higher degree of purification for both make-up and condensate return water. High steam purity requires extremely low limits on boiler water solids, sodium and silica(4). See the literature review on standards for more information on boiler water quality requirements.

Ultra purification methods generally include pretreatment, followed by membrane-based and ion exchange-based deionization (including electrodialysis and continuous deionization) and post treatment.

3.1 PRETREATMENT TECHNOLOGIES

Selection of a particular treatment will depend on the feed water quality, the downstream process and the purity specifications of the application. Manufacturers of reverse osmosis, ion exchange and distillation equipment often specify minimum purity levels to which the feed water must be pretreated (usually to achieve a particle size of less than 5 or 10 microns in diameter). Common pretreatment methods include[4] :

- cartridge filtration which consists of a cartridge with a porous, graded-density structure with a network of flow channels.

- sand filters of single, dual or multimedia type. Dual and multimedia filters are designed to allow coarse to fine gradation in the direction of water flow.

- softeners/scale control; the deposition of scale (e.g. $CaCO_3$, $CaSO_4$, SiO_2 deposits) is controlled by adding acid to the feed water, ion exchange softening or using scale inhibitors. The addition of acid converts carbonate to carbon dioxide which is not rejected by the reverse osmosis (RO) membrane. The water thus requires degasification to improve product quality. Dissolved silica will be rejected by RO membranes[14]. RO membranes and ion exchange resins will remove soluble Fe^{2+}, however, if Fe is oxidised to Fe^{3+}, insoluble $Fe(OH)_3$ will precipitate and foul the membrane or resin.

- activated carbon treatment is used to remove organics and total organic carbon (TOC) and can also be used to reduce chlorine contamination.

- organic scavenging adsorbents; anion exchange resins are used for the removal of organics in feed waters. The organics are removed through a combination of adsorption and ion exchange.

- colloids (charged and suspended submicron particles) removal through ultra filtration, flocculation with filtration, synthetic adsorbents and cartridge filtration. Colloids accumulate and form a fouling layer on membrane surfaces if they are not removed.

- microorganism removal by chlorination, microfiltration, ultrafiltration, UV radiation, heat, ozonation and special adsorbents.

3.2 DEIONIZATION PROCESSES

Once the water has been pre-treated, high purity water is produced via various deionization processes; reverse osmosis (RO), ion exchange, continuous deionization and electrodialysis[4].

Reverse osmosis is capable of removing ionic impurities, particulates and colloids, organics, micro-organisms and pyrogenic (refractory materials originating from high temperature processes) materials from water[15]. These ion exchange resins remove ions from water. Resins can be either cation-exchange resins or anion exchange resins. These resins are fouled by chlorine, iron, manganese, organics, micro-organisms and particles which must be removed by pretreatment.

Deionization by ion exchange occurs in either a separate bed or mixed bed process. In the separate bed process, cation resins and anion resins are loaded into separate pressure vessels. Water passes first through the cation resin, exchanging cations in the water for H^+, and then through the anion resin, where anions are exchanged for OH^-. Pure water is produced(4).

The separate bed process allows some salt leakage into the water through the cation resin. The mixed bed method (an intimate mix of the cation and anion resins) does not allow salt leakage since the salts created by contact with the cation resin are immediately neutralised by the anion resin(4).

Continuous deionization combines ion-exchange resins and membranes with electricity to deionize water. The basic repeating element consists of an ion-concentrating compartment and a diluting compartment bounded on each side by an anion-permeable and a cation-permeable resin. Deionization proceeds to purities associated with mixed bed ion exchange without chemical regeneration of the ion-exchange resins(4).

Electrodialysis removes ions by passing them through a stack of ion-permeable membranes that are impervious to water(16). The membranes are arranged as an array of alternate anionic and cationic membranes. Cationic membranes only allow positive ions to pass through, while anionic membranes only allow negative ions to pass through. A direct current electrical field drives the ions through the membranes. The elimination of suspended solids and the prevention of conditions favouring precipitation are essential in electrodialysis systems. The adsorption of suspended solids and precipitates on the surface of the membranes increase electrical resistance, resulting in membranic overheating and consequent membrane damage. Electrodialysis is widely used for the treatment of brackish or highly mineralised waters(4).

3.3 POST-TREATMENT METHODS

These methods target trace amounts of organics, particles, bacteria, dissolved gases and ions remaining in the waters purified by deionization.(4) Post-treatment options include :

- ultrafiltration for particle removal
- UV for bacteria control.
- UV - ozone in conjunction with deionization for organics removal.
- vacuum deaeration with catalytic reaction for eliminating dissolved gases.

In ultrafiltration, species are passed or rejected at the membrane surface on the basis of size(4,15). The pore sizes of an ultrafiltration membrane range from 0,001 to 0,02 microns.

Water purification by UV light is a rapid technique for eliminating bacteria without the use of heat or chemicals. A number of variables affect UV disinfection, including radiation intensity and water residence time. The drawback with UV treatment is that it cannot prevent any downstream contamination of organisms since a residual disinfectant does not remain in the water.

Membrane deaeration uses an oxygen-permeable membrane, which removes dissolved oxygen by allowing it to permeate the membrane into a low pressure vapour phase[4]. Ammonium reduction uses a palladium catalyst for the reaction between dissolved oxygen and ammonium to produce water and nitrogen. Nitrogen bubbling removes dissolved oxygen by providing a *counter current contact* between ultra pure water and nitrogen gas[4].

4. CONCLUSIONS

Various methods exist to treat mine service water for cooling water applications :

- scale protection may be achieved through the use of scale inhibitors, side stream softening, reverse osmosis, ion exchange and electrodialysis.
- fouling can be controlled by increasing the rate of blow down, side stream softening, the air rumbling procedure or chemical fouling inhibitors.
- microbiological growth can be controlled using oxidising biocides, non-oxidising biocides and biodispersants.
- corrosion and scaling protection may be achieved through the use of various chemical addition programs. Dirt deposits can be prevented by using coagulation, flocculation and sedimentation.

Boiler feed water undergoes three stages of treatment to produce pure water; pretreatment to achieve a particle size between 5 and 10 microns, deionization and postreatment.

Pretreatment methods include cartridge filtration, sand filtration, softening, activated carbon treatment and treatment with organic scavenging adsorbents. Pretreatment results in a high purity water which then undergoes reverse osmosis, ion exchange, continuous deionization or electrodialysis. Postreatment methods include ultrafiltration, UV treatment and vacuum deaeration.

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SEWAGE & SEWAGE SLUDGE TREATMENT

1. INTRODUCTION

In the past and under certain limited conditions it was possible to dispose of sewage without having to treat the flow. In this case disposal was by dilution into a larger body of water or through irrigation.

It is usually necessary, however, to treat the sewage to some degree before it can be disposed of or used in other applications. The degree of treatment required, depends on the end use and applicable legislation concerning final effluent discharge.

There are many methods, and variations thereof, used in the treatment of sewage. Most texts cover the whole spectrum of wastewater treatment from delivery through to disposal of both the effluent and solid waste(1, 2). The sewage may be treated to different degrees depending on the applicable legislation and certain aspects or unit processes may be omitted. The various unit operations involved in sewage treatment may be categorised according to the nature of the process, namely, physical, chemical or biological. The approach taken for this review is rather to categorise the treatment according to progressive stages in the complete treatment train. These various stages correlate to the three main objectives of sewage treatment:

- the removal of suspended and floating solid material (primary treatment),
- treatment of the biodegradable organic content (secondary treatment) and
- the elimination of pathogenic or disease causing organisms by disinfection.

The further treatment and disposal of the solid content is dealt with separately.

The choice of treatment process is dependant on the available technology at the time of plant design, the required degree of treatment and also on the capacity of the sewage works. Smaller sewage works, such as those serving mining communities, have particular operating requirements and difficulties. There are also certain advantages such as reduced volumes of sewage sludge requiring disposal. Various aspects of the operation of small sewage works are discussed in the literature(3, 4, 5, 6, 7, 8, 9).

2. PRIMARY TREATMENT

2.1 COARSE SUSPENDED SOLID REMOVAL

The removal of larger solid particles and floating matter is achieved with the use of screening devices of varying bar spacing. The screens require either a cleaning mechanism, such as a periodic water spray, or they should be a removable cage type for the purpose of periodic scraping. Coarse screens or racks are usually only used in larger installations and typically have a bar spacing greater than 25 mm. Medium screens, with a typical opening of 6-20 mm, are more common.

The disposal of the coarse and medium screenings can be achieved by burial, incineration or digestion but this is discussed more fully later. An alternative to coarse solid removal and disposal is the reduction of solid sizes, with the use of cutters or shredders. These will reduce the larger solids to a size which may pass through pumps for the eventual removal by sedimentation, for example. In some plants the screenings pass through a comminutor which shreds the particles to such a size that they may be returned to the main waste water flow for downstream treatment.

2.2 FINE SUSPENDED SOLID REMOVAL

The use of fine screens for removing fine suspended solids is very limited due to the low efficiencies possible and the preferred option is sedimentation. Sedimentation may either be used as a primary treatment, as is discussed in this section, or as a secondary treatment after aeration in the activated sludge process.

Sedimentation tanks are classified according to their design and function and the various types are discussed below.

2.2.1 Grit Chambers

In order to avoid damage to pumps and undesirable accumulation in downstream sedimentation tanks the grit and sand content of the sewage flow should be removed. A grit chamber should remove particles in excess of 0,2 mm and the chamber detention time should thus be sufficient to allow these particles to settle. The settling velocity of a 0,2 mm particle, of typical specific gravity 2,65, is approximately 25 mm/s at 20 °C. The more common horizontal flow, velocity controlled chambers are designed to maintain a flow velocity of approximately 3 m/s. Design must also consider the elimination of undesirable turbulence, fluctuations in sewage flow and the removal of accumulated grit.

2.2.2 Plain Sedimentation

In this type of simple sedimentation there is no chemical addition and sludge can be removed either periodically or continuously. The principle behind sedimentation is to provide a relatively quiescent state that allows the solid particles with a higher specific gravity than water to settle out. The design should also allow sufficient time for particle coalescence to occur. Usually, this detention time is of the order of 90-150 minutes and it is determined on the basis of a selected surface-loading rate. This figure is expressed in units of cubic metres per day per square metre of surface area. It is important to realise that the peak flow in small plants may be as high as 3 times the average flow. Sedimentation tanks can be rectangular or more commonly circular with a centre feed well and a rotating scraper mechanism for removal of the settled solids.

2.2.3 Other Methods

The process of flotation to remove suspended material has an advantage over conventional sedimentation in that it removes grease and other floatable material at the same time.

The improvement of screening materials and the development of fine screens has allowed for the replacement of sedimentation tanks by fine screens.

3. SECONDARY TREATMENT

The aim of secondary treatment is the oxidation of the putrescible content which remains after primary treatment. The treatment methods usually combine both physical and biological processes(10, 11). Physical filtration has been used to remove suspended solids as well as to reduce COD and nutrient concentrations(12).

There are numerous biological processes and variations thereof and they can be classified into aerobic, anaerobic or anoxic as well as suspended growth and attached growth depending on the characteristics of the micro-organisms.

3.1 AEROBIC PROCESSES

3.1.1 Activated sludge

The activated sludge process is a suspended growth process and it exists in many different forms. However, all are essentially similar and consist of a reactor containing the mixed liquor and a settling tank which separates the cells and treated waste water. The reactor is stirred continuously and this helps to maintain the aerobic environment. A portion of the settled cells are returned to the reactor as a recycle stream to maintain the required concentration of micro-organisms. The bacteria in the mixed liquor tank which may include *Pseudomonas, Zoogloea* and *Flavobacterium* use a portion of the organic waste as an energy source for the synthesis of new cells. The management and control of reactor conditions is particularly important for this process and the control of operating parameters such as oxygen uptake and CO_2 production have been investigated(13).

In some activated sludge plants the oxidation of ammonia to nitrate is carried out simultaneously to produce the stable nitrate state. This nitrification has the advantage of reducing the BOD demand of the final effluent. This is discussed more fully in Section 5.

3.1.2 Aerated lagoons

A variation of the activated sludge process is the aerated lagoon or pond where the pond acts as the reactor and oxygen is supplied by aerators. The use of water hyacinths in lagoons was investigated as a means of improving the effluent quality, to comply with stricter regulations(14).

Another suspended growth process is the stabilisation pond which consists of a pond containing both bacteria and algae. The algae assists with the supply of oxygen to maintain the aerobic conditions in the pond.

3.1.3 Trickling filter

This type of attached growth process consists of a bed, usually circular, of permeable media such as rock. Aerobic micro-organisms attach to the media and the waste water is percolated down through the bed. The organic material in the water is adsorbed onto the slime layer that forms on the media. This layer of slime increases in thickness as the micro-organisms grow. Any diffused oxygen is then consumed before it reaches the surface of the media and an anaerobic environment is created at this surface. Any adsorbed organic matter is metabolised before it reaches the micro-organisms at the media surface. Deprived of this carbon source, the micro-organisms enter the endogenous phase and begin to consume their own protoplasm. They cannot cling to the media and are consequently washed off by the trickling liquid. A sedimentation unit then removes the suspended solids content which has been washed off.

3.1.4 Rotating Biological Contactors

This apparatus consists of a series of closely spaced disks of polyethylene or polypropylene. As the disks rotate, the bacterial population attaches to the disk surface and forms a slime layer. The rotation of the disks is to allow the alternating contact of the micro-organisms with the organic material in the waste water and the oxygen in the air. In addition, the shearing force created by the rotation allows for the removal of excess solids from the disks. Similar disks have been used in the process of nitrification which is discussed further in Section 5.1(15).

3.2 ANAEROBIC PROCESSES

Anaerobic processes are not as widely used as aerobic processes for the treatment of waste water but there are limited applications(16) such as anaerobic digestion although, as discussed in Section 6.2, this is more widely used in the stabilisation of sewage sludge. Anaerobic ponds are used for the treatment of waste waters containing high solids concentrations. The ponds are usually deep and settling of the solids occurs simultaneously.

3.3 COMBINED AEROBIC/ANAEROBIC PROCESSES

The term facultative refers to organisms which have the ability to survive with or without free oxygen. Ponds which utilise a combination of aerobic and anaerobic conditions are thus called facultative ponds. Conditions near the surface of such ponds are aerobic and closer to the bottom facultative organisms are responsible for the biological activity.

Maturation ponds are usually used as a tertiary stage for the final polishing of secondary effluent.

4. DISINFECTION

The final stage in the treatment of the waste water before either discharge to the environment or reuse, is disinfection. The objective of waste water disinfection is to prevent the contamination, by pathogenic organisms, of receiving water bodies. These pathogenic or disease-causing organisms include *Salmonella typhosa* (typhoid fever), *Entamoeba histolytica* (dysentery), *Vibrio cholerae* (Cholera) amongst others. Generally, the enumeration of these pathogens is not carried out directly, due to the complexity of the analysis and the variety of analyses that would be required. The method most commonly used is to measure the quantity of certain indicator organisms such as total or faecal coliforms and to then extrapolate these results to pathogens.

There are various methods of disinfection, the most prevalent of which is chlorination. The ideal disinfectant should have a wide ranging biocidal efficiency at the applicable environmental conditions and should minimise the production of toxic by-products. Cost-effectiveness and ease of use are more obvious advantages.

The topic of disinfection is covered in a separate review and is therefore not considered further.

5. ADVANCED TREATMENT PROCESSES

The processes which are described in this section are aimed at removing or reducing the nitrogen and phosphorous concentrations in the waste water effluent before discharge to receiving water bodies. The reduction in these concentrations is becoming increasingly important in the light of better understanding of their impact and the resultant stricter applicable legislation(17, 18, 19, 20, 21, 22).

5.1 NITROGEN

Nitrogen exists in waste water in four forms: organic nitrogen, ammonia, nitrate and nitrite. Ammonia has a large oxygen demand and this is problematic when released to the environment. There are principally two strategies to reduce this potential problem i.e. conversion of the ammonia to nitrate before discharge or removal of the nitrogen content from the water.

The process of converting ammonia to nitrate is called nitrification and is achieved by both the aerobic suspended- and aerobic attached growth processes. Although nitrifying organisms are found in most aerobic processes such as activated sludge, the correct conditions need to be met in order to maintain sufficient populations. These would include additional oxygen, longer mean cell residence time and the maintenance of pH. Variations of the biofilm attachment media have been tested in an attempt to combine the aeration process with some degree of filtration and clarification[23]

For the removal of the nitrogen content, the nitrates need to be converted to nitrogen gas. This is achieved under anoxic (without oxygen) conditions and the use of deep bed sand filters for this purpose has been investigate[24].

5.2 PHOSPHOROUS

Phosphorous has been identified as the major contributor to the problem of eutrophication in water bodies, with the result that the Department of Water Affairs & Forestry imposed a special effluent standard of 1 mg/ℓ orthophosphate as P for certain sensitive catchments.

Phosphorous removal is complicated by the fact that almost all of the phosphorous content, after primary sedimentation, is soluble. Addition of chemicals such as lime and iron salts form insoluble precipitates. The addition can be carried out at various stages of treatment and is dependant on factors such as the influent phosphorous concentration, alkalinity and chemical cost.

6. TREATMENT OF SEWAGE SOLIDS

During the various waste water treatment processes, the solid content is removed in a variety of forms namely screenings, grit and sludge. This solid content is particularly problematic due to its offensive nature and the usually high water content. The further treatment of these solids is aimed mainly at reducing these problems through stabilisation and volume reduction for easier disposal. The various methods of sludge treatment are discussed and the final disposal thereof is discussed in the next section. The treatment of sewage sludge is largely dependant on the characteristics of the sludge which in turn is dependant on the upstream treatment processes and degree of digestion achieved.

6.1 THICKENING

There are various methods of sludge thickening or concentration but the majority involve gravity settling, although limited use is made of flotation and centrifugation. With relatively minor increases in solids content, a substantial reduction in sludge volume is achieved and this has obvious benefits for downstream processes such as digestion, drying and final disposal. These gravity thickeners are similar in design and operation to conventional slurry thickeners. They are designed on their hydraulic surface and solids loadings. The dilute sludge is fed via a centre well and after settling the thickened sludge is withdrawn from the tank bottom. This is stored and distributed when necessary to digesters or further dewatering processes. Flotation thickening is usually carried out by the dissolved-air method. In this case air is injected into a solution which is held at an elevated pressure. On depressurisation, fine bubbles are formed and the sludge is drawn to the surface for removal. This process is most efficient for sludge resulting from the activated sludge process.

6.2 STABILISATION

The aim of sludge stabilisation is to reduce the activity of microorganisms in the organic content of the sludge. A reduction in this microorganism activity leads to reductions in pathogen survival, release of offensive odours and putrefaction potential. The stabilisation can be achieved by a number of methods, namely, biological reduction or chemical oxidation of the volatile content, chemical addition and heat application aimed at sludge sterilisation.

Biological reduction can be achieved anaerobically or aerobically. The anaerobic process has historically been the more popular and involves the conversion of organic material to methane and carbon dioxide in an airtight vessel. The process can be carried out using a standard rate digester with a detention time of 30-60 days or by a high rate process where the contents are heated. The detention time is reduced to approximately 15 days.

The aerobic process has several advantages over the anaerobic route which include lower BOD content of the supernatant, production of a more manageable end product, fewer operational problems and lower capital cost. The cost of supplying oxygen and the fact that no methane is produced are disadvantages but these are less important for the smaller sewage works typical of mining plants. The aerobic digestion is similar to that of the activated sludge process where the microorganisms begin to consume their own protoplasm as the substrate supply is depleted.

Chemical oxidation can be achieved by the addition of chlorine to the sludge mixture. Chlorination is expensive and this technology is usually limited to treatment plants with a capacity smaller than about 0,2 m^3/s .

The addition of lime and the corresponding increase in pH creates an environment unsuitable for microorganisms but the organic content remains and the sludge has to be disposed of before re-infestation occurs.

The advantage of using heat as a stabilisation method is that it also assists in the conditioning of the sludge which improves the dewatering characteristics.

Thermal treatment of sludge is described in terms of it's benefit for downstream dewatering and denitrification[25].

6.3 SLUDGE CONDITIONING AND DEWATERING

Sludge is conditioned prior to final dewatering and this is achieved mainly by chemical addition or heat treatment. Chemical addition with ferric chloride or lime, for instance, causes the coagulation of solids and the release of absorbed water. Heat treatment also causes solids coagulation but the gel structure is destroyed and the water affinity of the solids is reduced.

Once conditioned, the sludge is ready for dewatering which is aimed at facilitating the final disposal. This is achieved by further reducing the sludge volume and producing a more manageable and less offensive product.

For smaller plants and particularly in the mining environment, where the availability of land is not limiting, the most common method of dewatering is with sludge drying beds. These beds should have an area of approximately 100 - 150 m^2 and the sludge is deposited to a depth of about 250 mm. The dewatering is achieved by evaporation and drainage below the bed. South African conditions allow the use of open beds and this sometimes causes odour problems. When dry, the sludge is cracked and dark in colour. It is then removed manually from the bed. Drying lagoons operate in a similar manner to that of beds but the depth of deposition is up to 1,25m and the cycle time is consequently much longer.

Other methods of dewatering include mechanical means such as vacuum, press or belt filtration[26], centrifugation as well as heat drying. These processes are more commonly used where there is insufficient land for a drying-bed application. It is suggested that in order to select the correct dewatering process, it is necessary to understand the nature of the moisture to solid bond in a particular sludge. These bonds may be chemical, physical or mechanical and are influenced differently by certain dewatering techniques.

7. DISPOSAL OF SEWAGE SOLIDS

The final disposal of the dried sewage sludge may be to a composting facility, landfill, waste disposal site, application to land, incineration or ocean disposal. A review of current legislation regarding these alternatives is presented by various authors[27]. It is clear that the application to land is an option which is potentially advantageous to both the farmer and the sludge producer. In South Africa as well as the UK, this method of disposal accounts for approximately half the total volume of sludge disposed[27, 28]. In the U.S.A. this method of disposal increased from 25 % in 1976 to 42 % in 1981.

Generally, sludge disposal is regulated partially under section 21 of the Water Act, 1956 since it arises from effluent treatment. It is also classified as a waste product requiring disposal and, as such, it is controlled by section 20 of the

Environment Conservation Act, 1989. However, the Department of National Health & Population Development (DNH&PD) have, in consultation with various organisations including the Departments of Water Affairs and Forestry , Agriculture and Environmental Affairs, drawn up a set of guidelines relating to the utilisation and disposal of sewage sludge on land. The guidelines make a sludge classification based on the potential to cause odour and fly nuisances as well as the ability to transmit pathogenic organisms(29). Type A, for example, is raw sludge or septic tank sludge which is usually unstable and poses a high health risk while type B is at least partially stabilised. Types C and D are stable products with upper limits on Faecal coliforms, Salmonella and metal concentrations (type D only). Various end-use options and limitations of each category are given in the guidelines. In terms of using sewage sludge for the stabilisation of mine dumps and subsequent grassing thereof, it is only Type A that is not permitted for this application. The use of types B-D is further restricted by qualifications regarding soil pH (>6,5) and the requirement for an adequate soil covering. A similar directive is available for EEC countries where there is increasing concern over the organic contaminants as well as toxic metal and pathogen concentrations.

The sludge composting facility which has been developed at the Johannesburg Council's Northern Wastewater Purification Works produced compost at a cost of R24,00/m^3 but concentrations of heavy metals limited the application thereof(30). A similar facility is planned for the southern Olifantsvlei works.

8. CONCLUSION

Sewage treatment and the disposal of the solid content is a complex operation which involves physical, chemical and biological unit operations. These processes can be combined in a variety of ways which is dependant on the capacity of the works, applicable legislation and the required degree of treatment, the nature of the sewage influent and the age of the treatment works.

Generally, sewage works serving mining populations have no restriction on available land and a relatively simple, maintenance-free system is preferable. In some cases, the operation of the sewage works is handled by specialised sub-contractors. Where possible, the treated effluent is used for irrigation on mine property, and increasingly, for slimes dam vegetation. The relatively low volumes of sewage sludge produced do not generally create disposal problems.

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POTABLE WATER TREATMENT

1. INTRODUCTION

Potable water refers to water of sufficiently high quality for drinking and food preparation. Water quality has an extremely wide spectrum of interpretation, and tends to be specific to the particular application of the water. Potable water quality is defined principally in texts prepared by municipal authorities [1, 2, 5, 6, 8] and in essence stipulate that potable water be free of microbial organisms capable of causing disease, and from minerals and organic substances which produce adverse physiological effects. To encourage consumption of healthy water it must be aesthetically acceptable, and thus be free of any apparent turbidity, colour, odour and any objectionable taste. The term potable water thus infers that it may be consumed in any desired volume without concern for adverse effects on health.

Water quality is a dynamic principle, and ever changing parameters, due to contamination and the influence of increasing populations and demand on water sources, constantly require adjustment of treatment technologies and water quality criteria. As a result water quality acquires increased significance, and a holistic approach to pollution control and purification within the context of the entire water cycle is essential.

2. CHARACTERISTICS AND CONTAMINANTS IN RAW WATER

Raw water sources are commonly classified as surface waters and ground waters. Surface water impoundments include rivers, lakes and reservoirs fed by rainfall and run-off, whereas ground waters originate from deep and shallow aquifers and springs fed by the slow percolation of surface waters through soil layers[1].

2.1 PHYSICAL AND ORGANOLEPTIC PROPERTIES OF RAW WATER

2.1.1 Electrical Conductivity

The electrical conductivity of water provides an indication of salinity or total dissolved salts (TDS) content. Low concentrations of salts particularly calcium and magnesium salts have nutritional value[2]. High concentrations of salts can have undesirable effects in terms of aesthetic quality and acceptability of water, health implications and economics. High salt concentrations adversely affect the kidneys, impart an unpleasant taste to water and intensify scaling and corrosion in the water distribution system[3, 4].

Electrical conductivity is an expression of the ability of water to conduct electrical current, resulting from the presence of charged species in solution. Various factors influence the electrical conductivity of an aqueous solution, and include the nature and concentration of the solutes present and their degree of ionic dissociation. Electrical conductivity is especially sensitive to variations in dissolved solids. Taste, hardness and corrosion are affected by the chlorides, sulphates, magnesium, calcium and carbonates in the water.[5] Correlations, norms and guideline limits together with treatment options are given in various texts[1, 2, 5].

2.1.2 Odour

Perceptions and sensations of odour and taste are complementary and are therefore difficult to differentiate. While the sense of smell is generally more sensitive than that of taste, the sense of taste is more useful in detecting inorganic constituents more effectively[5, 6]. Odour almost always indicates some form of pollution in the source of water or a problem during water purification and distribution.

The dominant causes of odour in raw waters relate to compounds produced by certain blue-green algae. The growth and metabolism of these micro-organisms is enhanced under conditions resulting from the disposal of raw sewage into water bodies. Sulphur compounds also contribute to odour in anoxic conditions where sulphates are converted to sulphides. Residual phenolic compounds and aminos can be converted to substances with strong odours during chlorine treatment (chlorophenols and dicloroaminos). The proliferation of iron and sulphur bacteria in water distribution systems may also be a source of odour after treatment[5]. Odours and tastes in water may be masked by high water hardness (TDS), but are exacerbated in soft waters. Odour elimination guidelines are presented in numerous sources.[1, 2, 3, 4, 6, 7, 8].

2.1.3 pH

The pH of most uncontaminated water sources is controlled by the carbon dioxide content and other minerals[1].The taste of water, the corrosivity, chlorination efficiency and the solubility of metal ions, etc., are all influenced by pH. Water at a low pH has a characteristic sour taste, while more alkaline waters have a soapy taste[4]. The pH of raw water also affects water treatment processes such as air stripping, chemical oxidation, chlorine disinfection, coagulation and chemical softening processes[2, 6]. Most raw water sources have pH values within the range 6,5-8,5[5], and are influenced by various factors including temperature, discharge of effluents, acidic mine drainage, run-off and organic decay processes.

2.1.4 Turbidity

Turbidity is a measure of light transmission through water. The turbidity of water is caused by the presence of suspended matter such as soil, sediments, humic organic substances and algae. Microorganisms are often associated with turbidity, hence low turbidity can minimise the potential for the transmission of infectious diseases[2]. Due to the association of microorganisms with turbidity, disinfectant dosages required to treat high turbidity waters are high. Turbidity is strongly associated with the colour and aesthetic quality of water[3, 5], and due to the complex and diverse composition of the constituents of turbidity, primary importance is attached to the removal of turbidity in potable water purification.

2.2 PHYSICAL AND CHEMICAL CONSTITUENTS OF RAW WATER

Minerals are found in all natural waters, and are referred to collectively as the total dissolved solids (TDS) content. The TDS typically include calcium, magnesium, sodium, chloride and sulphates. Silica and potassium contribute to a lesser extent to TDS.

Heavy metals which may be of interest in water supplies are arsenic, barium, cadmium, chromium, copper, iron, lead, manganese, mercury, selenium and zinc. These metals are normally only present in low concentrations due to the neutral pH of the water. Heavy metals are generally associated with severe toxicity and with potential bio-accumulation in the food chain. The occurrence of the contaminants in water is predominantly site specific and related to effluent discharges[2]. Other non-organic substances include aluminium which is found in natural water (aluminium salts used for coagulation also tend to leave residual aluminium in the water). Fluorides, nitrates, and radionuclides also occur to varying extents. The removal of most minerals is affected by the occurrence of organic or inorganic forms of the minerals[2]. Coagulation, flocculation, sedimentation and filtration effectively remove many contaminants[1].

Organics are broadly classified as natural organics and synthetic organics. Natural organics originate from soil run-off, aquatic biota, decomposition of vegetative biomass and human and animal wastes. Humic substances are important sources of turbidity. Fulvic acids, which commonly cause colour in raw water, are characterised by low molecular weights, these acids include most of the trihalomethane precursors. These organics which produce haloforms, principally form toxic trichloromethane during chlorine treatment of waters. Algal metabolites from blue-green algae such as cyanogosin, geosmin and methylisoborneol render water aesthetically unacceptable for potable use, and impose serious health problems.

Synthetic organics include pesticides and many other man-made compounds, predominantly chlorination byproducts. These substances contaminate surface waters through run-off and effluent disposal, and ground waters by leaching downwards from disposal sites and other areas of high concentration. These chemicals present toxicity hazards and removal problems, generally rendering the water unsuitable for treatment in terms of technology restrictions and economics.

2.3 MICROORGANISMS

Most raw water sources have microbiological populations composed of viruses, bacteria, protozoa, fungi, helminths and algae. Microbiologically polluted water constitutes an integral role in the transmission of infectious waterborne diseases. Microbiological pathogen detection levels are generally complex, and due to the large number of potential pathogens which may be associated with water, it is common practice to monitor and control microbiological water quality on the basis of levels of indicator organisms, rather than pathogenic organisms. Criteria for the selection of indicator organisms are given in the Canadian Water Quality Guidelines(6).

Faecal coliforms and more specifically *Escherichia coli*, are the most common bacterial indicators of faecal pollution, and hence of the possible presence of faecally associated pathogens. Numerous physical , physico-chemical and biochemical-biological factors influence the die-off of bacteria. Sunlight, temperature, salinity, competition, predatation, nutrient deficiencies, toxins, turbidity and organism density all influence the survival of bacteria. Extremes in pH, elevated temperature and solar radiation promote microbial decay, while high nutrient concentrations and lower temperatures promote microbial survival.(13, 14). Treatment of waters against bacterial contamination is governed by turbidity levels due to the association of the bacteria with particulate matter, in which they become adsorbed or enmeshed.(1, 2). High turbidity levels (above 30 NTU) generally require a complete treatment train involving full coagulation, flocculation, sedimentation and/or dissolved air flotation followed by filtration and disinfection.

Enteric viruses, are treated by the same processes employed for bacterial elimination, with treatment also focused on disinfection, while viral levels are ascertained by the presence of viral indicator organisms known as coliphages. Protozoan parasites such as *Giardo* and *Cryptosporidium* have several lifecycle stages of which the cysts are highly infectious and contribute significantly to human disease.(1, 2, 5, 6, 17, 18). Protozoans are highly resistant to chlorine and iodine(18,19), but can be removed by coagulation and flocculation, followed by complete filtration and disinfection. Ozonation and boiling of water also destroy cysts(6, 19). Additional information on the above micro-organisms is given in the literature review on microbiological contaminants.

3. POTABLE WATER TREATMENT

Water treatment processes provide barriers between consumers and waterborne disease. The principle reasons for treatment of surface and ground waters are to reduce particulates, reduce health risks from microbiological contaminants and to improve aesthetic values such as colour, odour and taste. In meeting these primary objectives, many organic and inorganic contaminants are also removed. The process generally includes multiple treatment techniques to effect water purification.

Common practice in water purification includes primary screening for the removal of large solids, pre-disinfection followed by chemical coagulant addition. Coagulation alters the physical and chemical state of dissolved and suspended solids, facilitating their removal by filtration. In conservative treatment schemes, a flocculation step is included which enhances particle agglomeration, improving removal efficiency. Sedimentation and filtration are utilised for further turbidity removal. In limited instances, enhanced purification of the water is achieved through ion exchange, softening and reverse osmosis. Activated carbon filtration and air stripping are also used for final organic removal. Final disinfection is employed for enhanced microbial elimination.

3.1 DISINFECTION

Disinfection is the most important process used in water treatment for the prevention of the spread of infectious diseases. Generally disinfection is accomplished through the use of an oxidant. Chlorine is the most widely used disinfectant, while chloramines, ozone and chlorine dioxide are also used. The discovery and subsequent regulation of certain chemical by-products of chlorination have increased the utilisation of other disinfectants such as ozone and chlorine dioxide. The selection and utilisation of disinfectants require an assessment of the reduction of health risks from infectious biological agents, increased toxicological risk due to residual levels of disinfectant at the point of use and the increased toxicological risks resulting from chemical by-products which are formed(1).

The most common locations for disinfection are at the inlet headworks to the water treatment plant and after final filtration. Pre-oxidation refers to the addition of an oxidant at the beginning of the treatment plant to remove odorous compounds and sulphides, to reduce coagulant demand, to reduce iron and manganese, and to prevent the formation of biological slimes and algae in downstream treatment processes. An alternative disinfection technique is UV light disinfection. Unlike chemical disinfection in general, UV does not depend on rate limiting diffusion of chemicals through water and the extracelluar polymeric layers which surround many of the microbes being disinfected.(20) The biocidal effects of UV light are essentially instantaneous, avoiding the need for large costly contact chambers. Unlike chlorination, UV does not produce chloro-organic by-products such as trichloromethanes which are carcinogenic(20). Unfortunately UV is less effective on turbid waters and is not generally used in South Africa. Ozone disinfection also provides an excellent predisinfection phase, with notable removal of iron and manganese as oxidised precipitates(21). The use of ozone and UV in predisinfection is developing favour in the USA(21).

Post-disinfection is usually considered to be the primary disinfection step, since the process is thought to be most reliable after all interfering particles have been removed by coagulation, flocculation, settling and filtration. In post-disinfection, the disinfectant is typically added immediately after final filtration. Chlorine and chloramines are sometimes added prior to filtration to prevent slime growth in the filter media.(1) In addition to primary disinfection, a secondary disinfectant is often added at the end of the treatment process to maintain a residual oxidant level in the distribution system. This practice is known as *residual maintenance*. Residuals of disinfectant prevent slime formation and the subsequent degradation of water quality in distribution piping. Generally the doses of disinfectant for post-disinfection are significantly less than those for predisinfection, since the oxidant demand has been reduced by the upstream processes.(1) The nature and byproduct formation of chemical disinfectants are discussed in various texts,(1, 2, 21) while UV disinfection is discussed by Cairns(20). Further discussion on the various disinfectants is presented in the literature review on disinfection.

3.2 COAGULATION AND FLOCCULATION

Coagulation involves the addition of chemicals to alter the chemical and physical state of dissolved and suspended solids to facilitate their removal by sedimentation and filtration. The most common coagulants are metal salts such as aluminium sulphate, ferrous sulphate and ferric chloride. Activated silicate (Na_2SiO_3) and organic polyelectrolytes are also used. These chemicals ensure aggregation of the suspended solids during the flocculation stage, which follows coagulation.

Coagulation may also remove some of the dissolved organic and inorganic compounds. The hydrolysing metal salts may react with the organic matter to form precipitates or they may form aluminium hydroxide or ferric hydroxide floc particles onto which the organic molecules adsorb. The organic particles are then removed by sedimentation or filtration. Adsorption and precipitation are also removal mechanisms.

Flocculation is a purely physical process in which water treated with coagulants is gently stirred to increase interparticle collisions and thus promote the formation of large particles. After flocculation most particulate aggregates will settle out during a period of one to two hours of sedimentation(1, 12). Additional information is given in the literature review on flocculation and settling.

The toxicity of coagulant residuals must be considered before selection and dosing, and detailed health effects and concentration dependence is discussed by the Drinking Water Health Effects Task Force(1) and WHO(5).

3.3 SEDIMENTATION AND FILTRATION

Sedimentation is a purely physical process like flocculation, and involves the separation of suspended solids from water by gravitational settling. The resulting effluent is then subjected to rapid media filtration to separate out solids which are still suspended in the water.

Rapid filters typically consist of 0,6 to 1,0 m of 0,5 to 1,0 mm sand and/or anthracite particles. Sand and anthracite coal are chosen as filtration media, since they are inert, and do not react with the water which passes through them. The operation of media filtration is dealt with in a separate literature review. Rapid filtration is effective in removing most of the particles which remain after sedimentation (1).

The substances which are removed by coagulation, flocculation, sedimentation and filtration accumulate as a sludge. The sludge must be disposed of in properly designed landfills or other acceptable solid waste management methods. The processes of coagulation, flocculation, sedimentation and filtration effectively remove many contaminants, with the reduction in turbidity being the most important. This treatment yields water of good clarity and enhances disinfection efficiency [1]. These processes are generally important for the removal of microorganisms, dissolved inorganic substances and certain metal ions.

3.4 ION EXCHANGE, SOFTENING AND REVERSE OSMOSIS

Ion exchange is a water purification process which involves the replacement of an ion or ion group in solution by another ion. The replacement ion is released into solution by a solid substrate, which then attracts displaced ions from solution. The process may exchange either cations or anions, depending on the properties of the solids matrix. The solid substrates are generally synthetic resins, designed to express a strong preference for the ions to be removed from solution, and a weak affinity for the ions to be exchanged. The efficiency of ion exchange resins depends on their chemical nature, porosity and mesh size. Stability is also important, if the mesh size of the resins are degraded by physical means, they may contaminate the water, however resins are normally designed not to contribute byproducts to the water.

Resins are inactivated by oxidation, and are poisoned by certain organic and inorganic compounds. The pores of the resin may be clogged by colloids suspended in the raw water, while the hydroxides of manganese and aluminium may precipitate on the surface of the resin beads resulting in deterioration. Effective pretreatment of the raw water by coagulation, flocculation, sedimentation and filtration prevents excessive resin deterioration. Ion exchange provides both positive and negative effects on water quality due to the nature of the process. While the process removes toxic metals and ionic contaminants, it adds other ions, principally sodium and aluminium which do constitute certain health risks [1, 2, 5, 7]. Ion exchange resins tend to adsorb natural organic compounds, such as humic acids (form trihalomethanes during chlorine disinfection), and may remove 50-90 % of all humic acids if applied prior to disinfection(1). However, lower molecular weight organic compounds are not removed.

Lime softening is a chemical process used to soften raw water, by the addition of lime (calcium hydroxide $Ca(OH)_2$ or lime soda ash (sodium carbonate $NaCO_3$) to assist precipitation of calcium as calcium carbonate ($CaCO_3$) and magnesium as magnesium hydroxide ($Mg(OH)_2$). High carbonate hardness waters, which have a high ratio of alkalinity to total hardness, may be treated with lime. When the alkalinity is lower than the total hardness, the difference is referred to as the non-carbonate hardness (NCH). The removal of NCH is achieved by the addition of either soda ash or caustic soda (NaOH). The only contaminants which may enter the process by lime softening are impurities which may occur in the lime soluble coagulants.

Reverse osmosis is essentially a physical process for water cleansing. Water is forced through a membrane at relatively high pressure, which filters out the larger ions and molecules. The membranes include cellulose acetate, polyamides and polysulfones. The principle use of reverse osmosis is in the desalination of water. Small amounts of chemicals are used to prevent scale formation on the semipermeable membrane, while sulphuric acid is added to neutralise carbonate alkalinity. Complexing agents are used to prevent the precipitation of calcium and magnesium salts. Sodium tripolyphosphate is added to improve water recovery and chlorine is added as a biocide when cellulose membranes are used.

Reverse osmosis is a highly effective process for the removal of the majority of high molecular weight organic compounds, including the precursors of trihalomethanes. Effectiveness of the process is governed by the membrane type, pressure and the ratio of product to reject water. The process of reverse osmosis is discussed in greater detail in a separate literature review.

3.5 ACTIVATED CARBON ADSORPTION AND AIR STRIPPING

Activated carbon adsorption and air stripping are used for the control of organic compound concentrations in water. Activated carbon is a large surface area adsorbent that can be used in powdered (PAC) or granular form (GAC). Granular activated carbon is most effective, and removes many types of organics including trihalomethanes and synthetic organic compounds. Radon is also effectively removed by GAC. GAC technology is specified as the best available technology for organic removal from water by the EPA(23). GAC particles range from 0,3 to 2,5 mm in diameter, and are used to treat water in a fixed bed, either as a rapid filter or as a separate contactor located after the rapid filter. The first is known as a filter adsorber, and the latter as a post-filter adsorber.

The removal efficiencies of GAC systems depend on many factors, including the utilisation of the filter as a filter adsorber or a post-filter adsorber, the quality of the water, adsorption characteristics of the organics in the water, the design of the filter and the frequency of replacement and regeneration of the GAC. Although GAC improves water quality by effectively removing organic compounds, GAC also provides an ideal environment for bacterial proliferation, since it chemically reduces chlorine.

Air stripping involves passing water down through a contactor, while air is injected upwards. Volatile organic compounds such as trichloroethylene and tetrachloroethylene are transferred from the water to the air due to their high volatility. Trihalomethanes may also be transferred. Radon is also removed by air stripping. The volume of air applied in this counter current configuration is an important factor in the removal efficiency of the air stripping process (1). The air stripping process is less expensive than GAC technology, and can be designed to remove up to 90 % of all volatile organic components from water streams.

4. CONCLUSION

The demand on natural waters for potable requirements are increasingly stringent due to the continuous development and refinement of analytical techniques, and the association of disease with newly discovered microbial agents and other raw water contaminants. The inevitable return of industrial effluent waters to their original water bodies always introduces higher levels of contaminants to the water system. The treatment routes for potable water have traditionally only included certain elements of the processes developed for potable water treatment, but as water demands on shared water resources increase with industrial growth and population development, the utilisation of more of these processes and consequently more complex treatment systems will become standard.

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PASSIVE TREATMENT

1. INTRODUCTION AND BACKGROUND

Since the introduction of the Environmental Management Programme Reports, the mining industry in general has become aware of the need to address the aspects of water quality management in a serious and organised manner. The principles of Best Available Technology Not Entailing Excessive Cost (BATNEEC) are applicable to the treatment of effluents and general water management. In addition to the treatment of effluents during mining activity, the provision of suitable management strategies for the post closure period needs to be addressed.

Presently, treatment strategies involve costly chemical and physical unit operations. The treatment processes generally require a high degree of maintenance and operating supervision. This approach is unsuitable for the post closure period, when resources, such as manpower and finances are limited or even non-existent. It is clear that the development of reliable treatment systems which are cheap and self-sustaining are an ideal solution for any industry. Such systems are known as passive treatment systems and have the following ideal characteristics: no requirement for mechanical energy input, minimum supervision and operational requirements and natural, prolonged and self-sustaining treatment materials. Due to the escalation in mining operating costs and increasing pressure on profits, the concept of passive treatment of poor quality water becomes even more attractive for the mining industry. In the case of mines due for closure, such treatment options are, in fact, essential if these defunct operations are not to become a burden to the state and consequently the tax paying public.

The possibility of using natural systems for treatment of polluted mine water was first explored approximately 25 years ago(1). It was noticed that naturally occurring *Sphagnum* wetlands had an ameliorating effect on local acid mine drainage. Several universities in conjunction with the United States Bureau of Mines initiated studies to explore the possibility of intentional use of such naturally occurring systems for mine effluent treatment. Problems experienced with the extended use of *Sphagnum* prompted the development of *Typha latifolia* (cattails) systems which exhibited improved tolerance to prolonged exposure to mine drainage as well as an ability to remove contaminant metals.

More modern systems have utilised a variety of plants, substrates and flow configurations. Pretreatment strategies such as neutralisation with limestone in anoxic beds have also been developed recently.

Conflicting results as to the success of constructed passive treatment facilities have been reported. Theoretical observations(2) are in contrast to the successful facilities implemented by authorities such as the Tennessee Valley

Authority(3). Hundreds of constructed wetlands have been implemented for various purposes and although many are relatively young systems, results are generally encouraging. It is likely that due to the fact that the technology is young and still developing many of the systems have been designed incorrectly. This aspect is currently receiving attention with the development of empirical relationships and publication of broad design guidelines.

2. OBJECTIVES

The focus of passive treatment in the South African mining environment is the amelioration of the acid mine drainage (AMD) which is characterised by low pH levels and elevated concentrations of sulphate and dissolved metals. The generation, migration and treatment of AMD is well documented and is also the subject of a separate literature review. Historically, treatment strategy has focused on the prevention at source, where possible, as well as chemical inhibition of the bacteria responsible for the catalysis of pyrite(FeS_2) oxidation.

Simply expressed, the pyrite oxidation reaction produces SO_4^{2+} and H^+ ions in the form of sulphuric acid. The increase in acidity and pH decrease has the effect of mobilising metals which are present in the ore or surrounding rock. These three pollution concerns, acidity, SO_4 and metal dissolution, are thus the focus of any passive treatment strategy. This review has been structured to discuss the treatment and available technology for each of the contaminant groups.

3. PASSIVE TREATMENT TECHNOLOGY

3.1 ACIDITY

The three different types of acidity include proton acidity which is an indication of the H^+ ion concentration, organic acidity which is associated with dissolved organic compounds and mineral acidity from dissolved metals such as Fe, Al and Mn due to their hydrolysis and H^+ ion production. Organic acidity is negligible in mine drainage waters but mineral acidity can be a significant contributor to the total acidity.

The two main methods of passively generating alkalinity are through the use of anoxic limestone drains (ALD) and compost wetlands.

3.1.1 Anoxic Limestone Drains (ALD)

The neutralisation of acidic mine drainage with these limestone beds is generally used as a pretreatment step prior to downstream sulphate reduction and metal removal processes. The ALD usually consists of a long narrow bed of limestone which is operated under anoxic (without oxygen) conditions. The bed is thus usually submerged and capped with a plastic lining material to prevent the ingress of oxygen. The capping also serves to trap released CO_2 which has the advantage of increasing the solubility of carbonate compounds. The gradient of the bed should be sufficient to overcome head losses within the bed. The ideal limestone has a high $CaCO_3$ content because of its higher reactivity and the most common particle size is approximately 70 mm

The factors which have limited the use and development of ALDs are the problems of limestone fouling and bed clogging. Water containing ferric iron (Fe^{3+}) and Al are particularly problematic due to the formation of the metal hydroxides, FeOOH and $Al(OH)_3$. In the presence of oxygen, ferrous iron (Fe^{2+}) also oxidises to ferric iron and the limestone fouling is worsened. The bed is kept anoxic to limit this oxidation step and maintain the iron content in the ferrous form. The ferric hydroxide precipitate chemically armours the limestone particles and prevents the further dissolution and production of alkalinity. Fouling due to the precipitation of the gelatinous aluminium hydroxide is not as harmful but can be problematic in terms of void plugging, leading to the eventual hydraulic failure of the bed. Although these limestone beds could be used in the short term to good effect in the removal of dissolved oxygen, iron and aluminium, sustained use will require the development of strategies to limit ferric hydroxide armouring and aluminium hydroxide clogging.

In terms of the iron precipitate, upstream compost beds may be installed which function as oxygen removal units. Effective use of these facilities ensures that the iron is maintained in the desired ferrous form. If the anoxic nature of the ALD is maintained, then the ferrous iron passes through the bed without any loss of limestone dissolution capability. Aeration at the outlet of the ALD, at the increased pH, will cause the iron to precipitate out of solution.

The management of the aluminium hydroxide clogging is more difficult since it cannot be controlled merely by eliminating the dissolved oxygen from the bed. One solution is the periodic flushing of the bed to remove the gelatinous material. Such a flushing mechanism would, however, need to be self-controlled and self-maintained for inclusion in a true passive system. The design of this passive flushing mechanism should be the subject of future studies. Recent developments include the possibility of using an upstream calcium fluoride bed for the precipitation of aluminium fluoride.

3.1.2 Compost wetland

An alternative, albeit more expensive, solution to alkalinity generation is the compost wetland which uses a combination of sulphate reducing bacterial activity and limestone dissolution. The wetlands commonly use mushroom compost which can have a high $CaCO_3$ content, although any compost mixture may be used. The addition of limestone to the compost increases the alkalinity generation. The compost bed may be stabilised by the introduction of emergent plants into the substrate which is typically 300 - 450 mm deep. These plants provide a further carbon source for the sulphate reduction process. Wetlands with subsurface flow have shown improvements in limestone dissolution as well as better winter performance.

3.2 METALS

The metals of concern in South African mining environments include iron, manganese and aluminium, of which iron is the most common. The most efficient passive method of removal of these metals is dependant on the overall water chemistry. In some cases, such as the sulphate reduction processes, the metal removal may be incidental. The various passive technologies are discussed below.

3.2.1 Chemical and precipitative methods

The most common methods of metal removal involve the precipitation of metal hydroxides under aerobic conditions, or in the case of the sulphate reduction process, the formation of insoluble metal sulphides.

3.2.1.1 Aerobic

Under aerobic conditions and in the presence of water, iron and manganese undergo the following hydrolysis reactions:

$$Fe^{2+} + \frac{1}{4} O_2 + \frac{3}{2} H_2O \rightarrow FeOOH + 2H^+ \qquad (1)$$

$$Fe^{3+} + 2H_2O \rightarrow FeOOH + 3H^+ \qquad (2)$$

$$Mn^{2+} + \frac{1}{4} O_2 + \frac{3}{2} H_2O \rightarrow MnOOH + 2H^+ \qquad (3)$$

Aluminium does not require the oxidation step and the hydroxide formation is given by equation (4)

$$Al^{3+} + 3H_2O \rightarrow Al(OH)_3 + 3H^+ \qquad (4)$$

Sufficient removal of these metals depends on the reaction kinetics and the detention time of the particular wetland system. pH influences both the reaction kinetics and the solubility of the metals in the water.

Ferrous iron may be oxidised abiotically or as a result of bacterial activity. The bacterial oxidation is dominant in the low pH range 2 -3 and abiotic oxidation is very fast at higher pH values (>8). Once the ferrous iron has been oxidised the hydrolysis reaction (2) takes place and the hydroxide precipitate is formed. The formation of H^+ ions has the effect of reducing pH, unless there is sufficient neutralising capacity. Aeration provides sufficient oxygen to oxidise 50-70 mg/l of ferrous iron to the ferric state. Concentrations higher than this level require the implementation of successive aeration-wetland (alkalinity producing) systems or SAPS.

The exact mechanism of Mn^{2+} precipitation under aerobic conditions is unclear. It is likely, however, that the manganese oxy-hydroxide shown in equation (3) would eventually oxidise to MnO_2. It is also possible, in alkaline waters, for Mn^{2+} to precipitate as a carbonate according to reaction(5).

$$Mn^{2+} + HCO_3^- \rightarrow MnCO_3 + H^+ \qquad (5)$$

Reactions (3) and (5) result in the production of H^+ and/or removal of alkalinity, HCO_3^-. The decrease in pH is not as marked as in the case of iron hydrolysis because the oxidation and hence hydrolysis does not occur significantly at pH values below 6.

The concentration of Al in water is primarily influenced by the pH and solubility of $Al(OH_3)$. Increases in pH due to carbonate dissolution or microbial activity in compost wetlands can cause the precipitation of $Al(OH_3)$. The hydroxide is highly soluble at pH values less than 4.

3.2.1.2 *Anaerobic*

Due to the absence of oxygen under anaerobic conditions, there is no oxidation of Fe^{2+} or Mn^{2+} . The bacterial reduction of sulphate is described by equation (7) in Section 3.3 and one of the products is hydrogen sulphide. This hydrogen sulphide reacts with the metal ions as shown in equation (6) to form the metal sulphide precipitates.

$$M^{2+} + H_2S + 2HCO_3^- \rightarrow MS + 2H_2O + 2CO_2 \qquad (6)$$

Aluminium does not form sulphide compounds under wetland conditions and the solubility of manganese sulphide is very high, with a solubility product of 5.6×10^{-16}. Although the solubility of iron sulphide is higher than that of copper, lead, and zinc, these other metals are not usually present in significant quantities. For the purpose of this discussion then, the hydrogen sulphide utilised in equation (6) has the most impact on dissolved iron concentrations. This precipitation process has the added benefit of reducing mineral acidity, although alkalinity is also consumed.

3.2.2 Biological methods

The potential of plants to remove metals through bioaccumulation has been identified. *Sphagnum* was initially identified as being able to accumulate iron but showed signs of petrification after prolonged exposure. *Typha* does not accumulate to the same toxic levels and is therefore more viable for use in longer term applications. Algal blooms have recently been associated with lowered manganese concentrations and both *Oscillatoria* and *Microspora* may be possibilities. The processes involved included both direct uptake and encrustation. However, the metal removal rates by plants are low compared to the precipitation methods and the fundamental role of plants in wetland systems is to stimulate microbial activity.

3.3 SULPHATES

The bacterial reduction and removal of sulphate is carried out under anaerobic conditions to produce hydrogen sulphide according to reaction (7)

$$2CH_2O + SO_4^{2-} \rightarrow H_2S + 2HCO_3^- \qquad (7)$$

The oxygen of the sulphate ion is used by sulphate reducing bacteria (SRB) to oxidise organic matter, represented as CH_2O. Hydrogen sulphide and the alkalinity generating bicarbonate, discussed in Section 3.1.2, are produced. The environmental conditions required by the SRB are as follows:

- presence of sulphate
- sufficient supply of carbon in a low molecular weight form such as lactate or acetate
- pH > 4 and redox potential < -200mV
- absence of oxidising agents (e.g. O_2, Fe^{3+},Mn^{4+})

The use of wetlands is ideal for SRB since the water acts as a barrier to oxygen movement and the required anaerobic environment is created. Two of the most common genera of SRB are *Desulphovibrio* and *Desulphotomaculum*.

It has been observed in some cases that the wetland acts as a sulphur source instead of a sink, as would be expected. In these cases elemental sulphur is oxidised to SO_4^{2-}. If the H_2S reaches an aerobic zone, such as the surface layer of the wetland, chemical oxidation back to the sulphate ion can occur and the wetland effectiveness is lost.

The emission of hydrogen sulphide and organic sulphide gases such as methyl sulphide and dimethyl sulphide has recently received attention in an attempt to determine the extent of sulphur release via the gas emission route. It is important to maximise the sulphide precipitation processes so that complete and efficient sulphate removal is achieved.

The process of sulphate reduction is dependant on the amount of organic matter available as well as the reactivity of the particular substrate. The rate of degradation for lignin is approximately 4-10 times slower than that of cellulose.

Historically, sulphate reducing wetlands have been constructed using a combination of limestone and mushroom compost which is a mixture of hay, horse manure, maize and wood chips. Most wetland systems have used *Typha* as the plant community, but a combination of *Typha capensis* (Bulrush) and *Phragmites australis* (Common reed) may be more suitable since the problem of seasonal variation in reduction rates may be alleviated by such a combination. The annual above ground dry mass production rate of these plants is approximately 4 kg/m². In terms of the self-propagation of any wetland this production rate needs to assessed with respect to the bacterial requirements.

There are no typical sulphate removal or sulphate reduction rates as conditions and design of various wetlands are extremely variable. Various studies have indicated a range of sulphate removal rates of between 0,18 - 25,5 g SO_4^{2-} /m^2.day. Many studies, however, yield removal rates of approximately 5 g SO_4^{2-} /m^2.day and this may be used as a rough guide of expected sulphate removal performance.

4. LONG TERM PERFORMANCE OF PASSIVE SYSTEMS

The long term performance of passive treatment systems is largely a matter of conjecture due to the fact that many systems are still relatively young. The mechanisms and technology of facilities such as anoxic limestone drains and wetlands are fairly well understood but are still the subject of exploratory studies. Design principles and guidelines are also constantly being revised as new information becomes available.

Certain qualitative aspects and potential problems of long term performance may be summarised as follows:

- Filling up of wetland systems with precipitated metals. Allowance for the eventual removal and disposal of these metal precipitates must be made. The possibility of metal recovery exists.
- Dissolution of limestone in ALD or compost beds. Allowance for the replacement of limestone must be made.
- Fouling and clogging of ALD. The fouling of limestone may be prevented but the removal of aluminium hydroxide precipitate implies a requirement for periodic flushing.
- Provision of sufficient and useful carbon for sulphate reducing bacteria in compost wetland. The wetland system must be designed and sized so as to meet the required long term carbon production rates. The carbon source should be self-sustaining and in the case of imminent mine closure, the passive system cannot rely on mine sources such as mine sewage works.
- The possibility of interruptions in flow needs to be considered. Where this is likely, adequate storage capacity should be provided.
- Integration of the wetland system into the surrounding environment.

These problems must be addressed in any long term passive system installation. Many, such as limestone dissolution and metal precipitate collection, may be quantified and the expected or planned lifetime of the passive system should then take these factors into account. The future replenishment of material and site maintenance would then occur simultaneously. Generally passive systems are designed to last approximately 20 - 30 years.

5. CONCLUSIONS

Although the technology of passive treatment for mine water is relatively young, the performance of many systems, particularly in the United States, has been encouraging. Mining related passive systems are aimed at acid neutralisation, dissolved metal recovery and sulphate removal, through bacterial reduction in compost wetlands. The concept is the focus of intense study as it offers a low cost, low maintenance solution to the treatment of poor quality mine drainage. This has particular benefits for the post closure period of mines when personnel and financial resources are no longer *available for the treatment of such water.*

Certain practical aspects such as fouling, life expectancy and wetland carbon source replenishment require further study but the chemistry and technology involved is fairly well understood. Improvements in sulphate removal rates and carbon utilisation will reduce the required wetland areas.

Presently, the extent of passive treatment in the South African mining industry is limited to three installations. In view of the EMPR requirements and the principles of BATNEEC, correct design and implementation of passive treatment facilities offer low cost solutions to treatment of both gold and coal mining effluents.

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EVAPORATION SYSTEMS

1. INTRODUCTION

Evaporation by its broadest definition is the conversion of a liquid to a vapour. In the narrower chemical engineering sense, evaporation is the removal of volatile solvent from a solution or a relatively dilute slurry by vaporizing the solvent. The transfer of heat is an important process in evaporation. Evaporation differs from distillation in that when the volatile steam consists of more than one component no attempt is made to separate these components(18).

The use of evaporation in the mining industry can be divided into two categories. Firstly, evaporation for concentration of wastes or brines (mainly solar evaporation and spray evaporation) and secondly for the treatment of brines or wastes to retrieve the water for reuse (by various types of evaporators) and for cooling. The latter category is referred to as mechanical or thermal evaporation.

Solar evaporation methods predate recorded history and probably still accounts for more total evaporation than any other method. Today, part of this is used as a means of disposing of effluents and waste brines(19).

South Africa is not richly endowed with abundant water resources and the problem is further exacerbated by an uneven geographical distribution of available water resources(1).

As is the case with most other industries, mining operations would not be possible without an adequate supply of water of the right quality. The gold mining industry uses water for a wide variety of purposes in its underground and surface operations(5).

Mine water quality varies significantly from mine to mine, but has generally deteriorated resulting in the Water Boards introducing water restrictions during the droughts in the 1980's, which caused the mines to place more emphasis on recycling and reuse of the water. The latter in turn, has led to an increase in the corrosion and scaling potential of the mine waters.

Although there are no exact figures on the cost of poor quality service water in mines, the most recent estimate is up to R300 million per annum for the gold mining industry as a whole. Recent developments such as the use of hydro-power, as well as possible future implementation of more restrictive legislation on the discharge of waste waters, have highlighted the poor quality of mine service waters currently in use in reticulation systems, and the need for improving their quality in order that they can be reused or safely disposed of(2).

Legislation pertinent to pollution makes it an offence to discharge water of the quality originating from, *inter alia*, slimes dams, or uranium plant raffinate(3).

The removal of soluble solids from waste streams is often the most difficult challenge facing waste process engineers. Because of the soluble and persistent nature of the substances in these streams the removal of water as opposed to the removal of the contaminant from the solution is required. Water removal from such streams normally implies that processes such as evaporation or membrane separation must be employed. In many cases, because of high concentration and the aggressive chemical nature of some of these substances, evaporation is virtually the only treatment method available for the reduction of volume of such effluent (4).

2. SOLAR EVAPORATION

Because phase change is the fundamental mechanism of separation in evaporation, it can achieve both the most condensed form of the unwanted solids as well as highest purity of the removed water. Evaporation is inherently an energy intensive process and comparatively high amounts of energy are required to vaporize water.

When a brine is involved it has a vapour pressure lower than that of water and loses proportionately more of the incoming energy to the atmosphere by convection instead of by evaporation. Thus the evaporation rate for saturated brine in large ponds is only about half that reported for fresh water in small pans. This can be of critical importance when there is substantial rainfall too, since rain falling on the ponds must be evaporated at the slower rate from pond brine(19).

Many gold mines, particularly those in the Free State, discharge their excess waste water into evaporation dams. The routes for water loss from these dams are through evaporation, seepage to ground water and occasional overflow to surface streams. The evaporation dams perform an essential water management function for those mines which experience large inflows of fossilized saline water which cannot be discharged to surface streams (5).

Evaporation dams are large surface area dams which are relatively shallow and contain large amounts of redundant water. The dams must be relatively shallow to increase the temperature of the water and consequently the evaporation rate(6).

A study done by Smith and Newell(7) on the simulation and economic evaluation of a solar evaporation system for concentrating sodium chloride brines showed that the concentration of brines for various processes could benefit from utilizing an open air solar evaporation process.

Water loss by evaporation on gold and uranium mines during cooling and chilling of service water needs be added into the water balance. A spray cooling tower or a spray pond may pre-cool warm service water coming from the main pump station (about 30 °C) to within about 2 °C to 2,5 °C of the ambient surface wet bulb temperature. This water is further chilled in a refrigeration plant to say 4 °C and is returned underground. Since the warm condenser outlet water also needs to be cooled, a second cooling tower is incorporated into the system. The loss of evaporation is assumed to be 2 % of the water flow rate per 10 °C cooling obtained in the tower. Loss of water droplets through wind is in the range of 1 to 5 % for the spray ponds; 0,3 to 1 % for the natural draught towers and 0,1 to 0,3 % for mechanical draught tower of the water flow per 10 °C cooling(22).

However, the primary consumptive use of water in the gold mining industry is in the tailings or slimes dams, return water and evaporation dams(9).

Most process water used in the metallurgical plants of the Free State gold mines is derived from reserves stored in shallow natural pans and dams. The strategy developed over the years was to dispose of excess water into evaporation pans during years of high rainfall. However, Gold Fields Water Board (GWB) introduced water restrictions in 1983(11). The Water Board now prohibits the use of open solar evaporation pans(20).

The management and prevention of water loss is now a priority in the mining industry. For example, a study by Haines(21) shows the water loss at Rustenburg Refiners. Various new methods were put into use to dispose of brines optimally. Not only evaporators were used, but rather a combination of Reverse Osmosis and various other water treatments.

Of the existing desalination technologies available, evaporative desalination or treatment processes are typically very expensive, and for this reason the cheaper membrane desalination processes are being considered and developed(10).

A study done in 1972(12) shows that two methods of disposal of brines have been utilized on a limited scale, and several others have been proposed. Evaporation of brines, primarily, by solar energy, in large water-tight ponds has been successfully employed, and numerous design studies have been based on this process. Improvements in the disposal of brines uses spray evaporation systems to optimise the concentration of the brines.

Solar radiation absorbed by these evaporation ponds supplies latent heat of evaporation, thus, allowing successive ponds to become progressively more concentrated, until the desired minerals precipitate. However, as the pond increases its temperature and slowly loses the capacity to evaporate, only a fraction of the solar energy input to an evaporating pond is converted into useful latent heat flux. Commercial production of useful minerals by evaporating ponds is thus inefficient in its use of solar energy, and is limited by the area available for the evaporating ponds.

2.1 SPRAY EVAPORATION SYSTEMS

Some preliminary tests of a spray pond evaporation system showed that pond areas could be substantially reduced (from that required if only solar evaporation is employed) if the brine is sprayed into the air through nozzles placed at intervals over the pond area. Rough cost estimates indicated a possibility that this system might offer a more economical way to dispose of brine (12).

Smoak(13) found that the spray delivered from the nozzle should expose a maximum water surface to evaporation. Depending on the relative size of the pond and spray systems, however, the brine spray droplets should not be so small that excessive driftage with wind occurs. Also, if a large percentage of the droplets completely evaporate while in the air, a fine salt powder will be left suspended and may be carried long distances by the wind.

An Enhanced Evaporation Spray System (EES) has been developed by ORMAT(14) for the concentration of aqueous solutions and recovery of minerals. The EES comprises a unique elevated low pressure spraying system which provides a large evaporation surface area by means of unisize falling droplets. This modular system solves the size problem of conventional evaporation ponds by requiring a land surface area which is six to ten times smaller than needed for conventional ponds.

The system utilises enthalpy from ambient air to evaporate water from liquid droplets. Specially selected nozzles, which produce a shower of droplets with desired diameter and having a very narrow size distribution, ensure a high evaporation rate and low carry over. Only a minimal number of large droplets, which evaporate less efficiently, and small droplets, which may be carried away by the wind, are present in the shower.

The rate of evaporation depends on droplet size, the height of the spraying system above the ground and the atmospheric conditions such as relative humidity and wind speed.

Consideration of the principle factors affecting the cost of brine concentration by spray and solar evaporation leads to the conclusion that spray evaporation will be cheaper in virtually all localities. Spray costs are high in cold humid climates, but in general, so are solar costs(12).

However, techniques need to be developed which are more effective than conventional evaporating ponds for evaporating water from brines without increasing the area dedicated to their production(8).

3. MECHANICAL/ THERMAL EVAPORATION

An evaporator is a device for removing solvent from a solution or a slurry by vaporizing the solvent. The feature distinguishing an evaporator from a drier is that the heat required for vaporizing the solvent is supplied to a liquid or a slurry rather than primarily to the solid itself. It differs from a distillation unit in that no attempt is generally made to separate components of the volatile phase. By far the largest number of evaporators are called on to remove only one solvent - water. The high latent heat of vaporization makes energy cost the most important consideration in evaporator selection and design. Evaporators can usually be arranged to reuse this latent heat a substantial number of times, thereby reducing the net energy cost and wastage. Evaporation has many applications in industry, including:

- Concentrating liquids for easier shipment or storage.
- Concentrating liquids in preparation for conversion to solids.
- Concentrating wastes to simplify ultimate disposal or to recover valuable constituents.
- Purifying liquids by salting out contaminants.
- Recovering purified solids from impure feeds.
- Recovering purified solvent, as for boiler feed water and in desalination of sea water (19).

Many types of evaporators are being used. These can be broadly classified as single stage evaporators, multistage evaporation, vapour recompression, natural circulation evaporators, forced circulation evaporators, film-type evaporators, multi-stage flash evaporation and fluid-bed evaporation (18) (17) (19). These will be briefly described, but the details of the operating mechanisms will be omitted.

3.1 SINGLE STAGE EVAPORATORS

Single stage evaporation can be accomplished by direct contact between combustion gases and the liquid being concentrated. Because of high cost and large space requirements, they are usually used only for highly corrosive feeds. A less expensive version is the submerged combustion evaporator, which has a gas or light burner submerged in a tank of the liquid. Combustion gases bubble through the liquid, providing excellent heat transfer without the risk of scaling. These evaporators have been used primarily for severely scaling liquids or corrosive liquids.

Most single stage evaporation is accomplished in the same types of indirect contact evaporators used in multiple stage systems. However, high energy consumption makes them economically unattractive for most services (19).

An example of a single stage evaporator is the "calendria"-type evaporator. The evaporator consists of a vertically arranged cylindrical vessel with two horizontal tube sheets built into the lower half. The tube sheets support vertical heat exchange tubes. Steam condensing on the outside of the tubes brings the liquid in the tubes to boiling. Vapour bubbles formed within the tubes discharge, mixed with liquid from the top of the tubes, promoting a natural recirculating fluid flow regime. A large vertical pipe is located in the centre of the tube bundle to provide for the necessary down flow of unvaporized solution.

The vapours produced separate from liquid in the upper section of the evaporator and, when necessary, flow through special internal separating devices to remove entrained liquid droplets. A condenser or preheater condenses the vapours to liquid(17).

3.2 MULTISTAGE EVAPORATION

Multistage evaporation is an effective means for reducing the steam consumption. A number of evaporator stages are connected together in such a manner that the solvent vapours generated in one stage serve as the heating medium for a succeeding stage. The vapours generated in the last stage must be condensed by a cooling medium, thus reducing cooling-water usage. Three fundamental variants of multistage evaporator arrangements are possible. The relative direction of flow of steam or solvent vapours versus that of the solution serves to differentiate them. The method most frequently used in practice is so-called cocurrent flow feeding. In this, the feed solution is pumped into the stage at highest pressure first. Then the solution flows from stage to stage one after another without pumping in the same direction as the solvent vapour using the stage to stage difference in pressure. The concentrated product leaves the system from the last, lowest pressure stage(17).

3.3 VAPOUR RECOMPRESSION EVAPORATION

Vapour recompression evaporation is a heat-pumping technique. A portion, or the total flow of solvent vapour leaving the evaporator vapour separation chamber, is compressed to a higher pressure and condensed in the steam chest of the same evaporator. In many cases, this processing scheme achieves specific heating steam efficiencies that can only be attained by multistage evaporator systems having as many as three to five stages. One may use steam jet ejectors operated by high pressure steam to compress the solvent vapours. This can also be done mechanically with centrifugal compressors(17).

Mechanical Vapour Compression is relatively expensive but is a rapid and continuous process producing a high quality product that can be returned to the system, thereby increasing the overall water recovery (10). The energy required by the process is only the mechanical power of the compressor, leading therefore to a low energy consumption. This is one of the main advantages of this type of system. There are other attractive features such as ease of operation and maintenance, reliability and resistance to corrosion, products of high purity distillate and economical operation(10).

3.4 NATURAL CIRCULATION EVAPORATORS

Natural circulation evaporators were the first developed commercially and still represent the largest number of units in operation. These evaporators utilise the density difference between the liquid and the generated vapour to circulate the liquid past the heating surface and thereby give good heat-transfer performance. The heat transfer tubes may be either vertical or horizontal, with the liquor either inside or outside the tubes. The horizontal tube type has the heating steam inside the tubes that are immersed in the boiling liquid. These evaporators are incorporated in the power plant cycle, usually as single effects heated by turbine bleed steam and exhausting vapour to the feed heater circuit.

Natural circulation evaporators can also be categorised under steam heated evaporators. Most industrial evaporation is accomplished in evaporators heated by steam and arranged to reuse the evolved vapour as the heating medium in the same or another part of the evaporator(19) (18).

3.5 FORCED CIRCULATION EVAPORATORS

The forced circulation evaporator, suitable for the largest variety of applications, is usually the most expensive type. It consists of a shell-and-tube heat exchanger, a vapour-liquid separator (variously called vapour head, vapour body, separator, flash chamber, or body) and a pump to circulate the liquor through the heater and back to the body. The system is usually arranged so that there is no boiling in the heater. The heat input is therefore absorbed as sensible heat, and vapour liberation does not occur until the liquor enters the flash chamber. Absorption of the heat input as sensible heat results in a temperature rise that reduces the net temperature difference available for heat transfer.

Forced circulation evaporators can be built for high single-unit capacities, having bodies as large as 15m in diameter. The circulating pump is usually the limiting factor and it is not unusual to provide a single body with as many as four or five separator heaters and circulating systems. For extreme fouling or scaling conditions, the individual heaters can be arranged to be valved from the evaporator and cleaned without interrupting system operation(18).

3.6 FILM-TYPE EVAPORATORS

In most evaporators it is the velocity of the thin liquid film near the tube wall that determines heat transfer performance. The principle function of the liquid in the core of the tubes is to drag this film past the heating surface. This liquid in the core, which comprises the bulk of the liquid circulated, requires energy to accelerate and decelerate it as it recirculates through the system. These energy requirements can be minimized by use of the film-type evaporators.

In the falling film evaporator, gravity rather than temperature difference provides the means of carrying the film past the heating surface. The "head loss" (one metre per metre of tube length) is much higher than in most other evaporators, resulting in high film velocities and excellent heat transfer performance. However, since it is only the fluid in the film that must be circulated, circulation volumes and energy requirements for circulation are much lower than those for conventional forced circulation evaporators. Flow of vapour in the core of the tubes is but little impeded by the liquid, so temperature differences and losses due to friction and acceleration are minimized. Tube proportions are generally about the same as in long tube vertical evaporators, so large amounts of heating can be packaged in single heating elements.

Vapour impurities are higher and vapour head diameters required are generally lower than for other types, even though in this type part of the diameter is occupied by the heating element and thus is not available for separation. The principal difficulty with falling film evaporators is securing uniform feed distribution to all the tubes(19).

3.7 MULTISTAGE FLASH EVAPORATION

Another means of reusing heat in an evaporator system is with the multistage flash (MSF) process. In this process a cool brine stream is heated progressively by vapour condensing at successively higher temperatures, and is then given a final boost in temperature by prime steam in a brine heater. The hot brine is then flashed down in stages to provide the vapour at successively lower temperatures that is the source heat for most of the brine heating in the heat recovery stages. Vapour condensed in heating the brine is also flashed down to the lower temperature stages and becomes the product of the system. The flashed brine from the last recovery stage may be discarded, or may be passed through a few heat reject stages which cools the brine enough so that it can condense vapour from the last recovery stage when it is recirculated.

The performance characteristics of a MSF evaporator are quite different from those of a multiple effect evaporator. In the latter the number of effects is the principal determinant of steam economy, and the amount of heating surface provided determines the evaporative capacity. In the MSF evaporator the capacity is determined only by the volume of brine circulated and the temperature range through which it flashes, while the steam economy is determined by the number of flash stages and the amount of heating surface used. Several times as many stages are needed in a MSF evaporator as effects in a multiple effect evaporator to achieve the same steam economy. The main advantages of the MSF system are that only one large recirculating pump is needed and that a large number of flash stages, including the associated surfaces, can be incorporated into one vessel (19).

3.8 FLUIDIZED BED EVAPORATION

At the heart of fluidized bed brine disposal is a pressure vessel containing hot salt particles fluidized by superheated steam. Fluidization causes the salt particles to move randomly in the fluidizing vessel, and the rapid particle mixing rates result in a nearly isothermal bed. Evaporation of the brine in the fluidized bed is accompanied by growth of a coating of solid salts on the bed particles. Heat is transferred to the bed from a submerged tube bundle. Vapour evolved from the bed is first scrubbed to remove particulate and is then condensed to recover heat and fresh water. Feed tubes with a large number of evenly spaced holes are required for even distribution of brine into the bed, otherwise the bed particles would agglomerate from the presence of excess liquid. Steam fluidization near the minimum possible level of superheat is required to economise on energy and in-bed heat transfer surface. Minimal corrosion, by virtue of the fluidized bed operation in a dry state with superheated steam, is an advantage over other methods of brine disposal. In addition, fluidized bed evaporation can be conducted at higher temperatures than is possible in other evaporative processes(16).

4. CONCLUSION

Many evaporation systems and evaporators are available for use in mining. The efficiency and cost of each option must be determined for each specific brine disposal facility or waste treatment facility. New techniques are being continuously developed providing a large range of options for selection in terms of optimum performance and economical value.

Water resource management practices are changing in South Africa and evaporation is not necessarily the optimum practice as it results in loss of valuable water. Accordingly, in the future it can be anticipated that greater emphasis will be placed on water treatment techniques which enable reclamation and reuse that is imperative to the conservation of water resources in South Africa. Some of these new techniques include Reverse Osmosis, Ion Exchange, Electrodialysis Reversal, Slurry Precipitation and Recycle Reverse Osmosis (SPARRO) to name a few.

Some case studies from the Water Research Commission are summarised below.

An in-house technical survey(23) conducted by Manganese Metals investigated the possibility of the application of a dozen or so established technologies for the treatment of their effluent. Using feedback obtained from commercial suppliers of waste water treatment plants the preferred methods of effluent treatment were ranked as follows:

i) lime precipitation followed by air stripping,
ii) electrodialysis and reverse osmosis,
iii) evaporation,
iv) biological denitrification,
v) ion exchange

Evaporation as a treatment method for the resultant effluent from manganese metal production involves a concentration step, it will therefore produce reusable water, but a concentrate will also be produced. The chemical loading in the original effluent and its disposal will be equally difficult.

Most power stations operate in a relatively simple manner and produce effluent. A combination of Reverse Osmosis, evaporation and drainage systems are applied in the treatment. Vapour compression evaporators are used[24].

Another example of where evaporation is used is at Sasol 2 and Sasol 3. Sasol uses a process of three RCC brine concentrator trains consisting of vertical falling film vapour compression evaporators. Heat transfer takes place across titanium tubes and the recycle process is of a seeded slurry nature[25].

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WATER AND SALT BALANCES

1. INTRODUCTION

Water and salt balances remain the most powerful potential tool available to mine water managers (12). Despite this fact, most mines do not have access to proper dynamic water balances which are updated on a regular basis to serve as management tools. The efficient management of water can reduce costs, it can reduce the volume of "lost" water and it can reduce the volume of water leaving the mine and purchased by the mine.

The water balance must identify all water circuits, across the mine, and ascribe accurate flow rates to each of the circuits. Once the water is in balance, salt concentrations can be added, to identify pollution sources (12).

Tables of flow and water quality data also have limited practical management value until they are converted to dynamic schematic representations. These schematic representations indicate the paths and volumes of water in circuit around the mine. Once the information has been collected, it can be placed on a computer spreadsheet programme. The use of a computer spreadsheet has the following benefits:

- Access to up-to-date and accurate water and salt balances at all times.
- Reduction in manpower required to prepare regular water and salt balances.
- Access to a powerful tool which assists mine personnel in managing their water systems, thereby reducing impact on the water environment and reducing wastage of valuable and costly water.
- Ability to audit water usage and wastage on the mine and instantly identify problem areas.
- Ability to provide accurate water balances when preparing EMPRs, when monitoring or auditing compliance with EMPR provisions and when entering into negotiations with water authorities.

2. WATER BALANCE

2.1 KEY COMPONENTS

The key components of a water balance are :

- accurate flow measurement of water in all the major circuits;
- accurate volumes of water in waste transported to waste residue deposits;
- accurate volumes of water entering the mine (either as purchased water or surface and subsurface flow);
- accurate volumes of water lost from the mine (from seepages, evaporation, evapotranspiration, discharge, cooling, treated sewage);

- accurate volumes of water in storage (surface and underground dams, residue deposits, settlers, etc.) (10, 11, 12).

Measurement of flow volumes on mines is dealt with in detail in the literature review on Monitoring and Instrumentation, but some of the more important aspects are highlighted here.

Water moves around mines in pipes, canals and open river systems. Additionally water may seep to ground, seep to surface or be lost in evaporation. The measurement of the volumes of water in these systems depends on the type of system.

2.1.1 Pipes

Water in pipes is measured mainly by insertion type and full bore type systems. Insertion type systems use a sensing element which is inserted through the pipe wall and into the fluid flow. The element is small in comparison to the pipe diameter and relatively small head losses are caused. An additional advantage with insertion type meters is the possibility of installing the instrument without the need for process downtime, as the element may be hot tapped into the flow line. Generally, insertion meters are less accurate than full-bore instruments but are usually cheaper and have a high degree of measurement repeatability.

Full-bore flowmeters are usually threaded into or flanged to an existing pipeline and have an inside diameter equal to that of the pipe. Many of the full-bore flowmeters utilise a flow restricting element and the measurement is by means of the differential pressure principle. The range of systems include concentric orifice plates, V-elements, venturi tubes, target meters and rotameters.

2.1.2 Open channels

Level measurement of water over specially designed and constructed open flow channel sections make it possible to calculate flowrate as a function of fluid depth. In many cases, fluid flow around a mine is by means of open channel and the flow measuring techniques developed for piped flow are thus unsuitable. Typically, a weir is constructed in the channel and water is forced to flow either over square shaped weirs, V-notch weirs or Parshall flumes. Standard equations are used to calculate the flow rate of water over the weir from details of the shape and dimension of the weir and the pond behind the weir and the depth of water at the weir.

2.1.3 Pump Hours

Routine maintenance of pumps is generally linked to the hours that a pump has worked. It therefore means that records are usually kept as to the number of hours that a pump is used. A popular method of determining flow rates, therefore is to convert the pump curve to pump hours to determine the volume of water pumped. This has a number of drawbacks. Wear and tear on the pump over time means that the original pump curve may not be accurate. Additionally, the pipe may scale and this would affect the actual volume pumped per hour.

2.1.4. Rivers and streams

The same principles apply for rivers and streams as for open channels. Weirs are normally constructed and depth readings are converted to flows using the discharge tables for the weir.

2.2 WATER CIRCUITS

The other main component for a water balance is an accurate schematic representation of the mine water circuits. This is needed to ensure that all appropriate flow volumes are measured, and that all circuits are included in the water balance.

The water circuits on a gold mine can be broadly divided into the following categories:

- general domestic circuits
- metallurgical plant circuits
- underground circuits
- slimes dam circuits
- environmental circuits (surface and groundwater)[12]

For coal mines the following circuits generally apply:

- general domestic circuits
- coal washing plant circuits
- mining circuits (underground and opencast)
- discard dump circuits
- environmental circuits (surface and groundwater)[12]

The development of the balance must take the form of input - output is equal to zero. In other words :

Total water in = Total water out

$$A + B + C + D = E + F + G + H$$

Flows which typically need to be taken into consideration when preparing flow balances are shown below.

2.2.1 Underground Systems

- Acid seepage
- Chilled water for cooling circuits
- Drinking water
- Mud from settlers
- Settled clear water
- Ventilation and evaporation losses
- Chilled mine service water
- Condensor circuit make-up
- Fissure water
- Ore/reef surface water
- Unchilled mine service water

2.2.2 Metallurgical Plants

- Chemical make-up water
- Evaporative circuits make-up water
- Mud from u/g settlers
- Plant area runoff
- Slimes dam return water
- Washdown water
- Evaporative circuits losses
- Gland service water
- Plant water
- Slimes/tailings discharge
- Surface moisture on ore
- Workshop water

2.2.3 Waste Deposits

- Evaporation
- Precipitation onto slimes dam or waste rock dump / residue dump
- Seepage to ground water
- Slimes/tailings discharged
- Surface water on waste rock
- Transpiration from vegetation
- Irrigation for vegetation
- Retained interstitial water
- Runoff from slimes dam/rock dump/residue dump
- Slimes dam / residue dump return water

2.2.4 Domestic Water Systems

- Hostel, villages, offices and plant potable supply
- Hostel, villages, offices and plant sewage return
- Potable supply from waterworks
- Treated sewage discharge
- Hostel, villages, offices and plant potable consumption
- Irrigation of gardens and sports fields
- Sewage sludge

2.2.5 Environmental Systems

- Diffuse source pollution to ground water
- Diffuse source effluents to surface water
- Point abstractions from surface water
- Wetland transpiration
- Borehole abstractions
- Evaporation from streams & dams
- Point source effluents
- Stream & dam bedlosses
- Surface runoff

Once the schematic has been developed, the routes can be converted to formulae. The formulae should be based where possible on actual measured volumes. If sufficient information is available, then all non-measured flow can be calculated from the balance. Assumptions regarding the apportionment of the various flows can be done, but care must be taken to validate the assumptions on a sound technical and scientific basis. It must be realised that if (n) formulae can be developed with (n) unknown variables, then the unknown flows can be solved by simultaneous solution of the formulae. If the required number of formulae cannot be developed, then the following actions can be taken:

- a flow monitoring programme can be started to measure the unknown flows;
- sound technical and scientific assumptions can be made to assign flow volumes to a unknown route.

If a water balance cannot be obtained for the particular circuit, then the first step would be to check the assumptions made. All factors concerning the assumptions will need to be tested critically. An example of such an error could be the volume of water derived from an old pump using the pump hours and the original pump curve. The pump may not be performing according to the original pump curve because of wear, scaling or corrosion. The whole development of an accurate water balance for a particular circuit is an iterative process. Each value should be rechecked once all the assumptions have been validated, in order to achieve a reliable water balance. It would be of benefit for all the assumptions and water balances to be checked by other knowledgeable mine staff, as this may clarify issues which affect the accuracy of the water balance.

Additional information on water use in the various circuits are presented in the following literature reviews:

- Water use and impacts
- Water reclamation
- Residue deposits
- Groundwater
- Stormwater management
- Sewage treatment
- Potable water treatment

2.3 WORKED EXAMPLE

In order to demonstrate how water balances are developed, a worked example is presented here(12). A typical gold mine is used as the example, but the principles would apply to a coal mine as well.

2.3.1 Surface water domestic circuits

A schematic diagram indicating the surface water domestic circuits is presented in Figure 1. The water balance for the domestic circuits is as follows (all figures are Mℓ/month):

(1) = (2) +(3) + (4) + (5) + (6) + (7) + (12) = (2) + (9) + (10) + (11) + (12) + (5) + x

where x represents a net consumption for the domestic system.

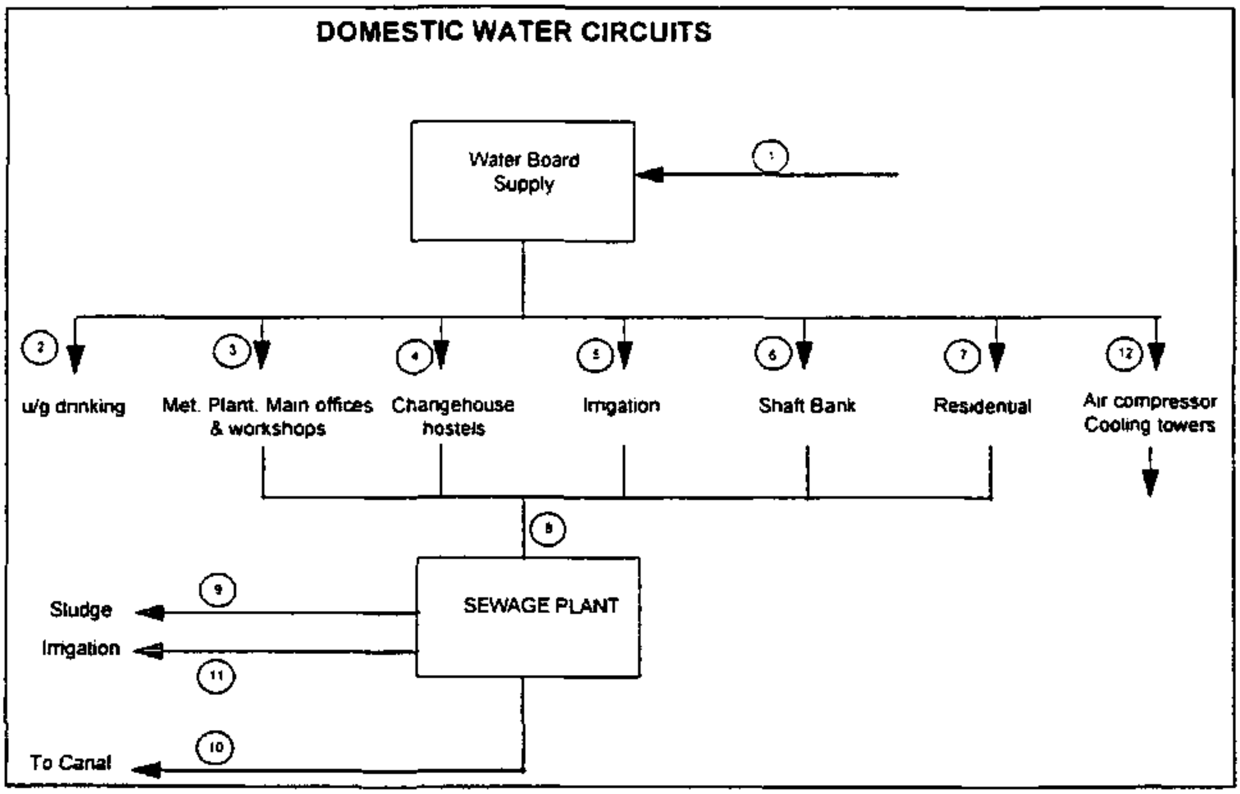

Figure 1 : General Surface Domestic Circuits.

This equates to:

In (185) = 6 + 14 + 59 + y + 18 + 51 + 1

In terms of the volume of water distributed to the various circuits, all are metered except the volume of water for irrigation. This is solved from the equation to give 36 Mℓ/month.

Out = 6 + z + 0,5 + 126,5 + 1 + 36 + x
Therefore x + z = 15 Mℓ/month

This calculation generates a net consumption loss for the system of 15 Mℓ /month. This figure is attributable to losses within the system, the portion of water within the sewage sludge, physical consumption, domestic garden irrigation, washing use and any discrepancies between the Water Board meter readings and the mine's own water meter readings.

2.3.2 General Water Circuit

A schematic water balance diagram for this section of the mine is presented in Figure 2.

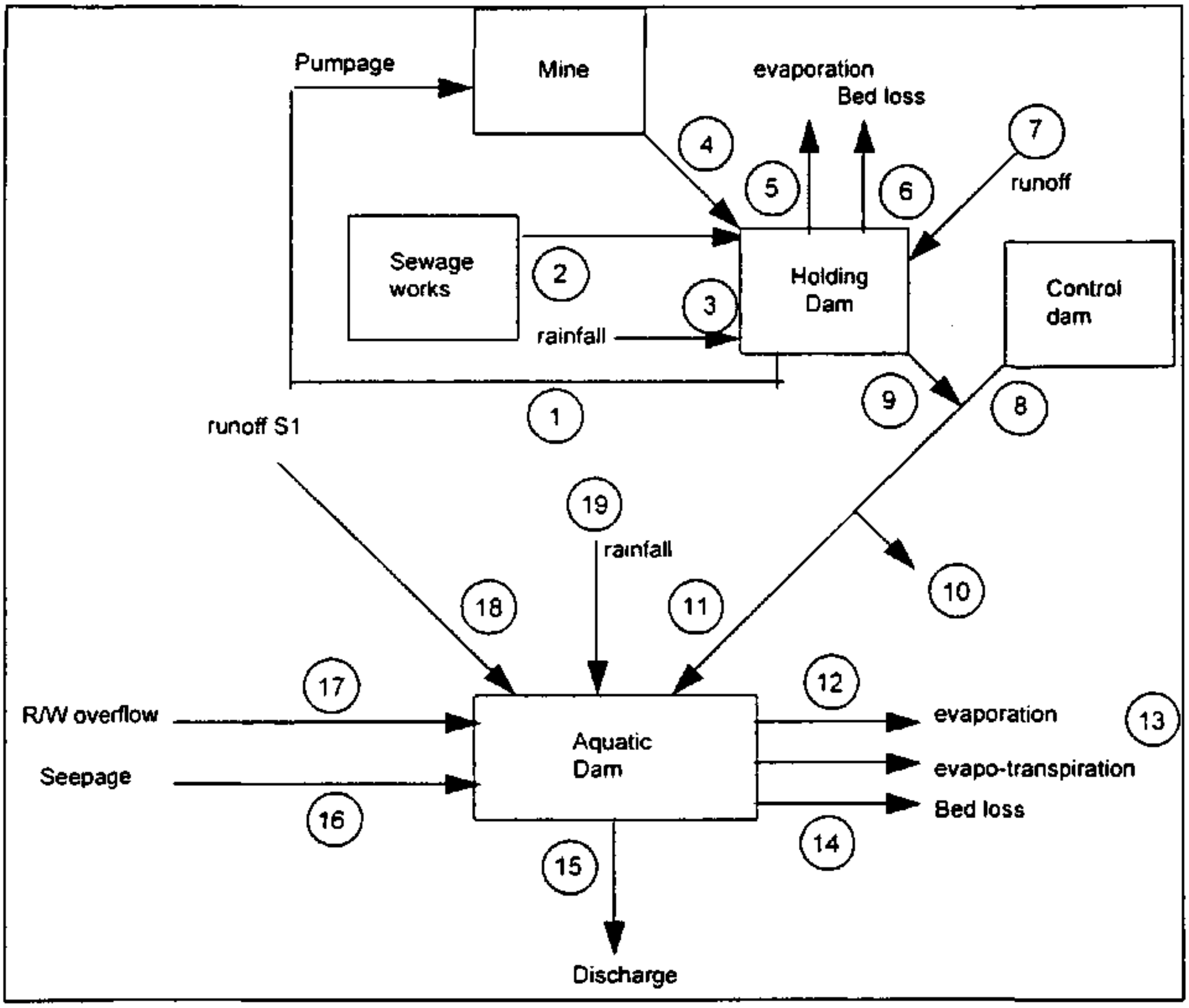

Figure 2 : Water Balance for General Water Circuits.

To derive the flow balance for this mine, two sections were identified:

- Holding dam adjacent to shaft areas
- Aquatic dam

2.3.2.1 *Holding dam adjacent to shafts*

The following equation was derived:

(2) + (3) + (4) + (7) = (1) + (5) + (6) + (9)

where :

Stream 1 Water pumped to Mine from adjacent mine

Flow meter readings for the mine pumpage were collected by mine personnel. Values of 78 Mℓ/month were supplied by the mine.

Stream 2 Discharges from the sewage works

The mine routinely monitors the volume of treated effluent which is discharged from the sewage works. The figure supplied was 90 Mℓ/month.

Stream 3 Rainfall

During the month, 6 mm of rain fell in this region. The volume of water input to the holding dam was calculated from the area of the dam multiplied by the rainfall. The following amount was calculated:

Direct rainfall on holding dam : 2000 m^2 x 0,006/1000 = 0,012 Mℓ

Stream 4 Discharges from Mine area

This water was not routinely measured and as such will be solved in the mass balance equation. A series of readings of actual flow volumes collected subsequently indicated that during this time approximately 174 Mℓ/month were being discharged.

Stream 5 Evaporation

The area of the holding dam is 2000 m^2. Average evaporation data for the month was supplied by the Weather Bureau, and the evaporation from the holding dam is calculated as follows:

Evaporation from holding dam : 2000 m^2 x 0,116/1000 = 0,232 Mℓ

Stream 6 Bedloss from holding dam

According to information obtained from the Department of Agriculture (4), the bedloss can be estimated to be equivalent to the direct rainfall on the system. However, other experience indicates that bedloss is typically much lower, i.e. 0,2 X direct rainfall. The bedloss factor could be dramatically affected by geological features such as faults or dykes underneath the channel bed or by recharge to dolomitic compartments. Because of the lack of precise measurements regarding bedloss, it is assumed that bedloss would be between 0,2 and 1,0 times the direct rainfall. Therefore an average of 0,6 is used for this water balance:

Bedloss from holding dam : 0,01 Mℓ.

Stream 7 Runoff

The runoff for this region was determined using the hydrological model, WRSM90. The rainfall was 6mm. As the area for which runoff needed to be calculated was too small for accurate modelling with WRSM90, the actual area was scaled up by a factor of 10 and the predicted runoff was divided by 10 to give the following:

Surface runoff to the holding dam : 0,94 Mℓ.

Stream 8 Overflow from Control dam

A number of actual flow measurements were taken from a disused Parshall flume which has been constructed downstream of the actual overflow of the dam, but in the main river channel. The flow readings were constant over the period of measurement:

Flow from Control dam : 3 Mℓ

Stream 9 Overflow from holding dam

The volume of water spilling from the holding dam was difficult to estimate. During one month, a number of actual flow measurements were taken, and these indicated diurnal variations within the system. No continuous monitoring equipment was available for this site.

It is therefore necessary to solve for this volume. A further equation was developed :

(9) = (11) - (8) + (10)

Stream 10 Evapotranspiration from wetlands around the holding dam

Evaporation from a small area of reeds between the flow measured downstream of the control dam but upstream of the culvert. This area was estimated to be 5600 m^2.

The evapotranspiration is generally calculated as the evaporation rate for the same surface area, multiplied by an evapotranspiration factor (E). Discussions with various researchers (2, 3) indicated that E may vary from 0,8 to 1,2 on the one hand and from 2 to 5 on the other hand. An extensive research programme undertaken during 1984 - 1987 (6) into rainfall recharge to ground water, concluded that " evapotranspiration remains a major unknown with regard to natural vegetation and needs to be addressed". Although this statement was made with regard to catchment evapotranspiration rather than for wetland systems, the reported range of E from 0,8 up to 5 would indicate that this remains a major unknown for wetland systems.

Experimental work undertaken for grasslands indicates that the evapotranspiration factor is affected by a wide range of variables, including wind speed, soil heat flux density, aerodynamic and canopy surface conductance, water vapour pressure, etc. which is best estimated by the Penman-Monteith equation (9).

Although it is reported that for North American conditions, E should generally be assumed to be 0,8, values of 1,8 have also been reported (8). Research carried out in India reported values of 1,6 - 1,7 for *Typha augustifolia L.* (9)and referred to previous research where E values as high as 2.5 were reported during the hottest months of the year.

It was decided to use the factor 1 for calculations of evapotranspiration at this mine.

Evapotranspiration : 5600 m^2 x 0,116/1000 = 0,65 Mℓ

Stream 11 Inflow to Aquatic Dam - East Stream

A point downstream of the discharge from the holding dam was equipped with gauge plates and readings were taken daily. The site where the gauge plates were installed was a culvert underneath the road. Standard engineering calibration equations (1, 5) and equations proposed by the British Standards Authority, indicated that such culverts are difficult to calibrate using general equations. It was therefore decided to calibrate the culverts using an impeller type flow meter. The following calibration curve was generated:

Volume = 0,29 + 1,23 x gauge plate reading

The integration of the converted gauge plate readings gave a volume of 139 Mℓ.

Stream 12 Overflow from holding dam - results

The solution to the equation proposed earlier is:

(9) = 139 - 3 + 0,65

Overflow from holding dam : 136,65 Mℓ

Results of Holding Dam Water Balance

The following equation is proposed:

(2) + (3) + (4) + (7) = (1) + (5) + (6) + (7) + (9)

(4) + 90 + 0,012 + 0,94 = 78 + 0,232 + 0,01 + 136,65
(4) + 90,952 = 214,9
(4) = 123,94

2.3.2.2 *Aquatic dam balance*

The following equation was derived:

(11) + (16) + (17) + (18) + (19) = (12) + (13) + (14) + (15)

Stream 13 Inflow from holding dam and control dam

As described earlier, this volume was measured via gauge plate readings taken off culverts to give a flow of 139 *Mℓ*/month.

Stream 14 Evaporation from Aquatic Dam

The evaporation from the aquatic dam is calculated from the area of the dam multiplied by the evaporation for the month.

Evaporation from Aquatic Dam : 110 000 m^2 x 0,116/1000 = 12,76 Mℓ/month

Stream 15 Evapotranspiration from Aquatic Dam

As discussed earlier, the evapotranspiration factor for this mine is 1. The evaporation is calculated from the area of wetlands and the evaporation for the month.

Evapotranspiration : 58940 m^2 x 0,116 / 1000 = 6,83 *Mℓ*.

Stream 16 Bedloss from Aquatic Dam

As discussed earlier, the bedloss is calculated at 60 percent of the direct rainfall on the system.

Bedloss from Aquatic Dam : 1,35 *Mℓ* x 0,6 = 0,81 *Mℓ*

Stream 17 Discharge from Aquatic Dam

The mine continuously discharges water from the Aquatic Dam for downstream users. Discharge is via a pipe running through the dam wall. A flow meter has been attached to this pipe and regular readings are taken by mine personnel. These readings were supplied by the mine. The metered readings for a 6 month period were averaged for a monthly flow. This equates to an average daily discharge of 4,8 m^3/minute (6912 m^3/day). The result would therefore be the following:

Discharge from Aquatic Dam : 209,5 *Mℓ*.

Stream 18 + 19 Seepage/overflow from slimes dam and return water dam

The exact volume of water seeping and overflowing into the Aquatic Dam from the slimes dam and the return water dam is unknown. It will therefore need to be solved in this equation.

Stream 20 Inflow from other major tributary - West Stream

The second major tributary which enters the Aquatic Dam is gauged in a similar manner to location (9). The same procedure for calibration was used. This yielded the following calibration equation:

Flow = 0,075 + 0,23 x gauge plate readings = 16 Mℓ/month

Stream 21 Rainfall

The direct rainfall on the aquatic dam and the wetlands which surround it is calculated from the area of the dam (110 000 m^2), the area of the wetland (58 940 m^2) and the rainfall of 6 mm. The areas of the dams were obtained from plans at the mine.

Direct rainfall on Aquatic Dam : 168 940 x 0,006/1000 = 1,01 Mℓ/month

Results of Aquatic Dam Water Balance

The following equation was derived:

(11) + (16) + (17) + (18) + (19) = (12) + (13) + (14) + (15)
139 + (16)+ (17) + 16 + 1,01 = 12,76 + 6,83 + 0,81 + 209,5
(16) + (17) + 156,01 = 229,9
(16) + (17) = 73,9 Mℓ

The equation is solved for the unknown volumes of seepages from the slimes dam entering the aquatic dam.

Point to Note

Unknown volumes of water at the scale of 74 Mℓ/month should be investigated if the water balance indicates that such volumes are entering the system from unmeasured sources.

2.3.3 Metallurgical Plant Circuits

The water balance for the metallurgical plant circuits is divided into three sections:

- metallurgical plant
- return water dam
- operational slimes dam

The schematic water balance is shown in Figure 3.

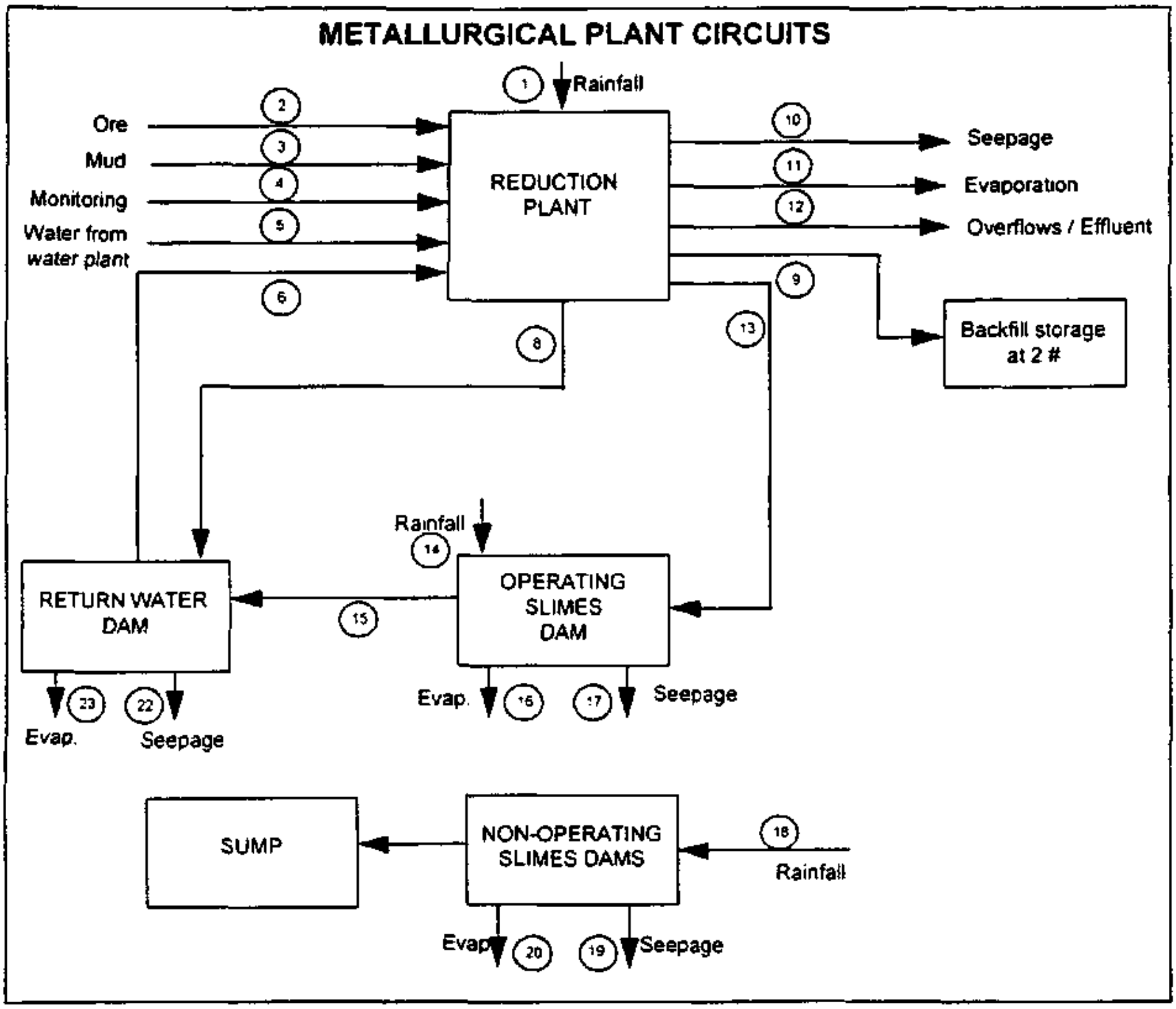

Figure 3 : Schematic Plan Of Metallurgical Plant Water Circuits.

2.3.3.1 *Metallurgical plant circuit*

The following equation has been defined:
(1) + (2) + (3) + (4) + (5) + (6) = (8) + (9) + (10) + (11) + (12) + (13)

where:
(2), (3), (4), (5), (6), (8), (9), (13) were supplied by the mine.
Rainfall figures were calculated to be 53 mm/month.
Runoff for the plant area was calculated as being 90 % of the rainfall
Evaporation is calculated.

The results of this equation are (all values are Mℓ/month):

(13) = (1) + (2) + (3) + (4) + (5) + (6) - (8) - (9) - (10) - (11) - (12)
(13) = 11 + 9 + 10 + 13 + 96 + 208 - 66 - 1 - x - y - z
(13) = 347 - 67 - (x + y + z)
(13) = 280 - (x + y + z)

The actual value recorded for water leaving the plant to the slimes dam (265Mℓ/month) is calculated from the volume of slimes pumped to the slimes dam. This would mean that $(x + y + z)$ has a value of 15 Mℓ/month, which can be attributed to seepage, evaporation, overflows and any other losses from the system.

2.3.3.2 *Return water dam circuit*

The following equation has been derived:

(15) + (8) = (6) + (22) + (23)

The results for this equation are (all values are Mℓ/month):

$147 + 66 = 208 + x + 0{,}5$
$213 = 208{,}5 + x$
$x = 4{,}5$

The volume of water lost to seepage is therefore calculated as 4,5 Mℓ/month.

2.3.3.3 *Operational slimes dam circuit*

The following equation has been derived:

(15) + (17) + (16) = (13) + (14)

The results of this equation are (all values are Mℓ/month):

$147 + 46 + x = 265 + 28$
$x = 100$

These results indicate that a volume 100 Mℓ/month of water is being lost from the slimes dam and return water dams via seepage.

Point to Note

The figures for rainfall and evaporation which were used were a monthly average. The effect of this is that the results of this water balance would only be applicable for a short period of time during the year. During the dry winter period, the volume of water in the slimes dams could be expected to drop as a result of decreased rainfall. During summer it could be expected that the increased rainfall would lead to an elevated water table in the slimes dam, and thus an increased amount of water seeping from the dam. The water balance for the disused slimes dams would also be dependent on the time of year and also the intensity of the rainfall.

2.3.4 Underground water circuits.

The underground water circuits are shown in Figure 4. The underground water balance equation can be divided into two sections: fissure water and reused water for mining operations, and the settler circuits.

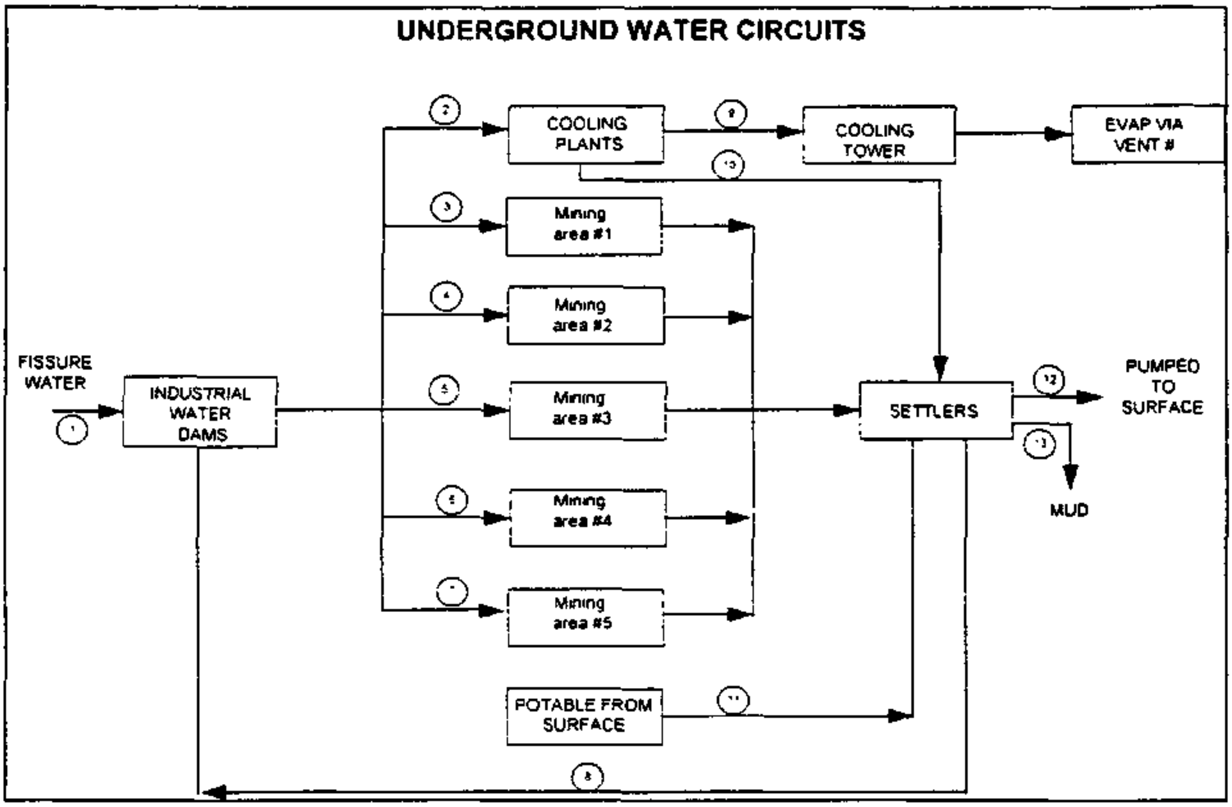

Figure 4 : Schematic Plan Of Underground Water Circuits

The equation for the mining operations is :

(1) + (8) = (2) + (3) + (4) + (5) + (6) + (7)

(1) = (2) + (3) + (4) + (5) + (6) + (7) - (8)

The equation will be solved for the estimated contribution of fissure water to the underground water circuits. The results for this equation are (all values are Mℓ/month):

(1) = 20 + 60 + 180 + 150 + 150 + 270 -300
(1) = 530 Mℓ/month

The volume of water used in the cooling plants is measured. The other values are estimates based on mining activities in the various sections of the mine.

The equation for the underground settlers is:

(8) + (12) + (13) = (10) + (3) + (4) + (5) + (6) + (7) + (11) - x

where x represents the net consumption within the water circuits.

The volume of water pumped from underground is measured. All other values are estimates based on activities in the various sections of the mine. The results for the equation are (all values are Mℓ/month):

$$\begin{aligned} 300 + 533 + 10 &= 11 + 60 + 180 + 150 + 150 + 270 + 6 - x \\ 843 &= 827 - x \\ x &= -18 \text{ Mℓ/month} \end{aligned}$$

This negative value indicates that more water is leaving the system than entering it. The reason for this is that the figures supplied for one or more of the circuits is incorrect. The values used in the water balances for underground are based on mining operations in the various levels of the mine and are estimates. The actual measured volume pumped to surface is 533 Mℓ. A volume of 9 Mℓ/month is sent to the cooling tower.

Point to Note:

The accuracy of all flow volumes is important. In the underground circuits, the discrepancy indicates that at least one of the volumes calculated is incorrect.

3. SALT BALANCE

3.1 KEY COMPONENTS

The key requirements for the calculation of a salt balance are the following:

- an accurate water balance for the relevant mine water circuit;
- total dissolved salt (or another conservative element) concentrations for each of the major routes in the circuit;
- a summary diagram indicating in and out circuits.

The basic principle in calculating the salt balance is held in the calculation of loads. The calculation is :

flow x salt concentration = salt load

Salt loadings can only be calculated for conservative salts such as sulphate, chloride, sodium and TDS. Non-conservative elements such as calcium, phosphate, nitrate, heavy metals and other compounds which are biologically metabolised or chemically precipitated, cannot be used for the salt balance unless factors can be applied to account for the metabolic and precipitated losses.

3.2 GENERATING THE SALT BALANCE

The same basic structure as for the water balance is applicable to generating the salt balance. Each of the input streams for which water flow volumes are known must have a salt concentration, and each of the outflows from the system must have a salt concentration. This can be presented schematically :

$$A + B + C + D = E + F + G + H$$

$$f_A.c_A + f_B.c_B + f_C.c_C + f_D.c_D = f_E.c_E + f_F.c_F + f_G.c_G + f_H.c_H$$

where f = flow and c = concentration

The salt balance can also be used to test the water balance. Typically, the quantification of salts in the water is done fairly accurately in a laboratory or at the very least, a similar error exists in all the values determined. If a salt balance cannot be generated, then this would indicate that one of the water circuits may be the problem. A close inspection of the loads from each of the routes would tend to point towards the flow which has the least reliable data. These principles in generating the salt balance are best illustrated by way of example. The Aquatic dam in the general water circuits water balance example is used to illustrate this. Figure 5 depicts the summarised input and output diagram.

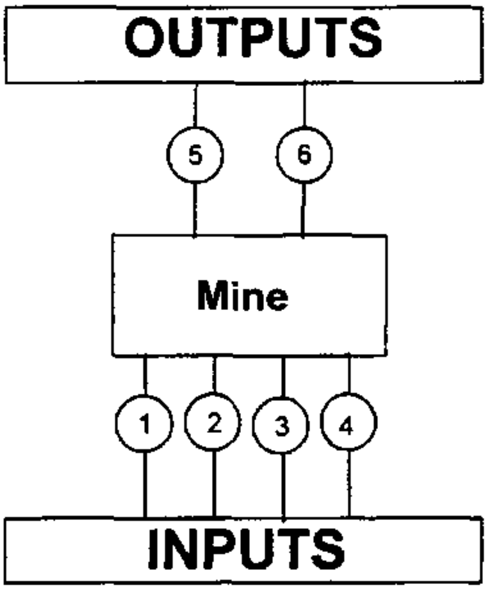

Figure 5 : Summary Water Balance.

Table 1 presents summarised TDS data for the streams at the Aquatic Dam.

Table 1 : TDS Data for Aquatic Dam

STREAM	TDS (mg/l)
East Stream	357
West Stream	561
Rainfall	15
Seeps/ return water overflow	650
Discharge	514
Bedloss	514

The following equation and resultant loads were derived:

Input = (357 x 139) + (15 x 1,01) + (561 x 16) + (650 x 73,9)
= 106 861 kg

Output = (514 x 209.5) + (514 x 0.81)
= 108 099 kg

The net difference between the two represents 1 %. This indicates that the assumptions made in the generation of the water balance are valid.

In this example, the salt balance showed a very small difference between the input loading and the output loading. If the difference had been significant however, it could indicate the following:

- one of the flows for one of the circuits is incorrect. For example, if the volume associated with the seepages had been 50 percent less, i.e. 37 Mℓ, then the net difference between the input loads and output loads would be 25 000 kg, or 23 per cent. This would indicate that the volume of water associated with the seeps was incorrect, and the water balance would have to be redone.

- one or more of the flows was missing. For example, if the salt balance indicated that there was 23 per cent more salt load leaving the system than entering, it would tend to indicate that there was another source or sources of water entering the system. The use of the water and salt balance could therefore provide an indication as to whether seepages or other pollution sources are entering the surface water systems. If there was a higher load entering the system than leaving it, it could identify whether an abstraction from the system has been overlooked.

4. WATER AND SALT BALANCES IN MINE WATER MANAGEMENT

The worked examples above, have indicated the methods used to calculate the water balance, and to generate the salt balance. Some of the points to note are:

- If all major flows are measured (both in and out) then losses from the system can be calculated;
- If the water balance indicates that a considerable volume of water is entering the system unguaged, then management action can be put into place to investigate the source, nature, and management of the water;
- If the water balance is not valid. i.e. there is a discrepancy of in / out and there is a negative number generated, this indicates that flow volumes for one or more circuits are inaccurate;
- If the water balance is not done dynamically, i.e. on a routine ongoing basis, discrepancies between dry and wet years can provide erroneous values.

In terms of the salt balance, detailed analysis indicates that in the above example, approximately 45 per cent of the salt load is entering the system from seepages from the slimes dams. This clearly identifies this route as a significant pollution source, and indicates that management action is required to address this issue. Such management actions have been identified and are presented in the manual to assess and ameliorate the impact of gold mining operations on the surface water environment(12).

Thus, water and salt balances, if done accurately and properly, can indicate where the mine's water losses and pollution sources are. Once these sources are ranked according to significance, then appropriate management strategies can be used to address these sources. Not only can water and salt balances indicate the contributions to pollution from diffuse sources, but it can also be used to rationalise incorrect use of high quality water (which has a cost implication to the mine), and it can also be used to show where clean and dirty water systems can be separated.

As mentioned at the beginning of this review, water and salt balances provide a useful management tool for the mine, but they should be seen as an intricate part of mine water management and as such should be updated regularly and routinely. The optimum time frame would be weekly updates, which would allow the manager to keep control of water use and movement around the mine. Such a system would best be handled in a computerised water and salt balance programme, where the time to update the system is reduced and the programme automatically calculates all facets of the water and salt balance.

Additionally, a reliable water and salt balance will allow the mine to test what-if scenarios with the mine water systems, before actual construction takes place. It allows the manager to test different conditions, and different options with one, two or more circuits, and the manager will be able to see the effect all over the mine.

The water and salt balance is not a finite set of numbers and formulae. Operational changes within the mine may affect the assumptions made in the generation of the water balance. Additionally, the reliability of meters, pumps and weir measurements may change over time, thus increasing the uncertainty in the volumes of water measured. To address this issue, annual audits would critically assess each of the circuits and routes to check whether deviation has occurred, and would allow the mine to recalibrate the water and salt balance.

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STORM WATER MANAGEMENT ON MINES

1. INTRODUCTION

When planning a stormwater management programme, there is a strong tendency to consider only the quantity of stormwater runoff and its effects such as flooding, and to ignore the quality impacts. The long-term effects of polluted stormwater, however, can be very significant, and in some cases may be the most critical aspect of stormwater management(1).

Stormwater management is one of the significant aspects to be considered in mine water management. Runoff waters from mine areas may contain pollutants and if not controlled, may contaminate ground and surface water. Any release of this water to surface streams could adversely effect the quality of these streams and rivers (2).

2. GENERAL HYDROLOGY

Storm hydrology applies hydrological science to the calculations necessary for predicting storm runoff, with emphasis on the hydrological effects of land development and mining. Storm management is concerned with several aspects of hydrology. Hydrology has as its objective, the analysis and prediction of movements of water in and under the ground, but it is inescapably linked to meteorology, because the data on precipitation are much more extensive and more susceptible to statistical analysis than the data on water movement.(1)

Stormwater management requires a careful analysis of expected rainfall duration, intensity, and distribution in time and space. It is necessary either to study the effects of a large number of storms that actually happened or to use statistics compiled from studies of many storms in an area of concern. Modern statistical hydrology tends towards the use of actual storm data when computers and models are used. Other approaches often use a design storm derived from the statistical data. With such great variations of rainfall in time and space, it has become necessary to average rainfall quantities for use in analysis. Rainfall is averaged both in time and space. As an example the 100-year, 60-minute map can be used. The 100-year event has a 1 % probability of occurrence each year. Design storms of all kinds can be prepared from these rainfall maps and publications. Once rainfall maps are available, the depth-duration-frequency curve is a useful format for employing the data(1)

2.1 VARIABLES AFFECTING RUNOFF

- *Rainfall intensities* : Runoff from a surface is strongly dependent not only on the permeability of the surface layers but also on the intensity of the rain. If the surface is already soaked, the runoff becomes substantial. These relationships resort under the domain of hydrology. Rainfall/runoff relationship is important for predicting stormwater runoff(5).

- *Slope* : Erosion on slopes is a function of the eroding power of raindrops, running water, and sliding or flowing earth masses, and of the erodibility of the soil or rock(6). In almost all environments, except very flat plains, the development of a watershed brings with it an erosional disequilibrium, with increased bed load carrying capacity sufficient to initiate a channel erosion and channel enlargement process of long duration, called channel degradation. Special channel management measures may also be used to reduce effective stream slope at flood stages(1).

- *Vegetation* : During a single rainstorm upon previously dry plants a high initial loss is caused by evaporation but, as the plant surfaces become moist, losses decline to a lower level that is controlled by the water content or humidity of the air. Interception losses for a short duration rainstorm of a few hours can vary from 10-20 per cent on crops to 5-50 per cent on forests(6).

- *Soil* : Infiltration is the process by which water enters the surface horizon of the soil. Porosity and permeability are the two most important factors. The first one determines the void space in the soil through which water can pass(6).

These variables have a clear effect on runoff and, therefore, stormwater. In gold and coal fields these variables must be used as the basis from which prediction and management must be done.

2.2 RUNOFF CALCULATION

The calculation of runoff and therefore stormwater, involves passing the estimated rainfall through a transformation to arrive at estimated runoff. There are numerous approaches that can be used, ranging from simple empirical approaches to very elaborate methods.

- The rational method : This method yields only a peak flow, not a hydrograph. The rational method assumes that runoff is linearly related to rainfall and therefore assumes that a 10-year rain produces a 10-year runoff.

- Soil *conservation service method (SCS) : The methods are applicable to* small and large catchments. When the entire hydrogram is desired, it is necessary to perform a watershed routing, using the subcatchment's runoff histogram and unit hydrograph. (SCS the tabular method hydrographs which result from computer calculated watershed routings).

- The synthetic unit hydrograph approach : For watersheds larger than 200 *acres the "Colorado Urban Hydrograph Procedure" (CUHP) method can be* used. It is simply a version of the Snyder Unit Hydrograph Approach and can be implemented very simply without a computer(1).

Because of the need for storage-capacity data, land surveys were combined with aerial-photographic mapping techniques to determine pertinent preflood and maximum -flood water-surface levels, develop topographic maps, and determine storage -capacity data[7]. By using these methods to calculate the stormwater volumes/flows, predictions can be made which are useful in planning and designing preventative measures.

2.3 HYDROLOGICAL MODELS

Stormwater models and (Geographic Information Systems) GIS may be used to predict and characterise the local or regional hydrology, facilitating preventative measure decisions. The principal use of models is in situations where direct measurements are either impossible or impractical[1].

- *Surface Runoff Modelling* : Surface runoff, as the term is generally used in stormwater modelling, includes both surface and some subsurface runoff elements. Most surface runoff models simulate flow as a series of simple hydrologic/hydraulic processes. Flow routing is accomplished by Manning's equation or a similar technique(1).

- *Transport System Modelling* : The design and analysis of large trunk sewers, storm drains, and interceptor sewers involve more sophisticated hydraulics than were considered in surface runoff computation(1).

- *Surface Runoff Quality Modelling* : A number of stormwater models have attached to them some kind of water quality routine. Many problems concerning water quality can be solved by modelling only the flow. Surface runoff quality modelling is not nearly as advanced as is hydrologic/hydraulic modelling(1).

- *SWMM* (Storm Water Management Model) : This is a package of models linked by an executive program(1).

- *GIS for practical hydrology* : The coupling of GIS and hydrologic models seems to be a logical direction, as the applications compliment each other. The input parameters for existing hydrologic models are prepared in the GIS and passed on to the model through an interface. There could be interactions between GIS and a hydrologic model at various stages of the modelling if the model produces spatial output. Hydrologic modelling in a GIS is generally limited to time averaged values, whereby the temporal variations have been accumulated or averaged out. The creation of a digital elevation model (DEM) from contour data or stereo-satellite data may be regarded as a typical GIS facility. The results of hydrologic modelling may be displayed in a GIS and merged with other geographic data, for planning and decision-making.

In the model-driven approach, a model or set of models is defined and thus the required spatial input for the preparation of the input data and output maps. The data-driven approach limits the input spatial data to parameters that can be obtained from generally available maps.

GIS, hydrology and resource management are a good combination in putting data in a format with which they are familiar. The assumptions are that the selected model produces reliable results and the spatial parameters can be adequately obtained by survey or measurement. An important function of the GIS is to present the results of hydrologic studies in a format that is understood by authorities that have to act upon the results[9].

- *Remotely sensed data* : For understanding of the hydrology of areas with little available data, a better insight into the distribution of the physical characteristics of the catchments is needed. By image processing techniques, images can be produced which depict some of the characteristics, notably the cover types such as areas with dense vegetation, water bodies and areas with bare soils or outcrops. The vegetation pattern influences run off and evaporation.

Weather satellites and rainfall depth are handy in predicting floods. METEOSAT is used for its high temporal resolution required for connective rainfall. Imaging in the visible (VIS) part of the spectrum is used for determining cloud development and some aspects of cloud thickness.[9]

The ACRU hydrological modelling system is a multi purpose model, outputting either runoff elements like stormflow, baseflow, peak discharge at daily, monthly or annual level or reservoir yield analysis, or sediment yield analysis, soil water status and total evaporation etc.

It is a physical-conceptual model, i.e. it is conceptual in that it conceives of a one, two or three dimensional system in which important processes and couplings are idealised, and physical to the degree that physical processes are represented explicitly.

Inputs required include, location, catchment, climate, land use, soils, hydrology ,etc., which are entered into the model and via operational modes of *moisture budgeting/evapotranspiration modelling* to point vs. *lumped* vs. distributed vs. GIS modes give simulation options.

Another mathematical model, to simulate the movement of water through an interlinked system of catchments, river reaches and reservoirs, is WRSM90. It is a modular system with a runoff module, a channel reach module, a reservoir module and an irrigation module.

A schematic that represents the river system is the basis of the model, and the modules are connected by means of routes. Eventually volume outputs from one system or set of routes can be used as an input into another set.

Mapping from dynamic water bodies can be assessed quite accurately from the high resolution imagery. The remotely sensed inundation limits provide a unique way of checking the predictions of a hydrologic flood model. Satellite imagery has been used in a court of law to provide evidence of the origin of flood waters.[9]

- *Creation of flood maps* : Using the DEM, a map can be created showing the extent of the area that is inundated, provided that the DEM is accurate enough. By comparing this result flood map with the classified image an assessment of the quality of the DEM can be made. To find the topographical connected areas a topographic operator of a GIS will have to be used in conjunction with a connectivity operator.

The SPOT multispectral imagery is used to detect water bodies. By using the simple technique of density slicing it was possible to differentiate between flooded and non-flooded areas. These flood maps can be calculated.[9]

3. OBJECTIVES OF STORMWATER MANAGEMENT

Stormwater management consists of using the array of approaches available for addressing mine drainage and flood problems on the smaller streams. In predicting floods the assumption is made that the cycle of floods of the past would be repeated in the future, with no essential change in frequency and severity. Stormwater management essentially deals with very localised problems and focuses on small streams and headwaters. Flooding of mines can be very counter productive with delayed ore extraction, costs of pumping the water back to the surface or even the loss of lives[10].

Mining activities lead to increased runoff from surfaces that are polluted, thus eventually leading to the pollution of rivers if not managed properly. Mining demands a high assurance of water supply, using almost 3 per cent of all water used and may have negative impacts on rivers[10]. Mining effluents, often contain toxic chemical compounds. Moreover, seepage from gold and especially coal mines may lead to an increase in the salinity of nearby rivers.

Landfill has a capacity to generate leachate and particular attention must be paid to the protection of surface and ground waters. This polluted soil and groundwater can contaminate stormwater flowing over it and transporting the polluted sediment[12]. The quality of stormwater runoff can be greatly affected by the type of development within the catchment and the level of management of the engineering services[13].

The principle applied is that discharges must be controlled at their source[14]. So far as the quality of surface waters is concerned, it is vital that equity be achieved. his requires a quantity-quality relationship throughout the catchment area, controlled abstractions, conservation of resources and intelligent inter-basin transfers. Catchment areas can be divided into low-, medium- and high damage potential areas where stormwater runoff is concerned[15]. Mines must have a 100 year stormwater level and consider their catchment areas according to it[16].

3.1 STORMWATER ON OPEN CAST MINES

- *Strip mining* : This involves the removing and stockpiling of soil, drilling and blasting the rock above the coal seam, removing the blasted overburden by draglines(14). During the mining operation, considerable volumes of groundwater may be encountered and rainwater also falls onto the pit and spoil piles, thereby creating a potential for water pollution(19). This potential is controlled by installing separate clean and dirty water collection circuits. Clean water running off unmined and rehabilitated land is channelled, where possible, into nearby streams. Dirty water from the pit, haul roads and plant areas is collected and reused for activities that do not require good quality water such as dust control and coal washing(20).

- *Open-pit mining* : The ore body is blasted and followed deeper into the ground in a series of benches. The waste rock, and waste that is separated from the ore during processing, is dumped on the surface away from the pit. The objectives for land use following open-pit mining are more limited than for strip mining, because there is usually insufficient waste or even tailings to fill the pit. The main objective is usually to make the pit walls safe and to landscape the waste rock dumps, but many innovative solutions have been used. These include using the pit as a waste disposal site, filling it with water with the intention of creating an ultimate recreation/water supply/nature conservation end-use or simply fencing it in and leaving it as a tourist attraction(21). Therefore stormwater runoff can flow into these newly created dams.

3.2 SHAFT FLOOD PREVENTION

- *Shallow underground mining* : Near surface aquifers may break, draining water into the collapsed workings. Rain falling on these areas goes into the workings, adding to water load. This water becomes acidic and high in salt concentration(22).

- *Deep underground mining* : The mine dewatering can cause dewatering of previously water-filled caverns below the water table and erosion of fine soil material out of the overlying strata causing instability in the strata which in turns collapses, resulting in sinkholes. The collapse is sudden and it is difficult to predict when and where it will happen(23). The sinkholes allow surface water and rainfall to flow directly into the workings, which can significantly increase pumping costs. The slime can then be covered with soil up to the level of the original surface and the area revegetated. If the water table is re-established, the danger of further sinkholes occurring is limited(14).

3.3 SLIMES DAM EROSION PREVENTION

The design of the outer walls of residue dams is dependent on a number of operational factors as well as considerations relating to ultimate closure or rehabilitation of the tailings' deposits. If space is not a problem it may be possible to contemplate slopes as flat as 15° to 20°. This will not only reduce erosion of material by water runoff down the slope, but will also simplify vegetating the slope and will reduce maintenance after closure.

- *Dump reclamation* : Dump reclamation refers to the processing of these dumps. The material in the old dump is monitored into a sluice. The sluice gravitates the dump material to a low point where it is collected and pumped, through a pipeline, to the treatment plant that could be located some distance away. The main environmental protection activities during reclamation are to keep stormwater away from the working areas, to prevent dust pollution during dry, windy conditions. In many cases, the top of the soil profile at these sites has been contaminated with acid water seepage from the dump. This may be more offensive due to an increased pyrite concentration(20).

The following information is necessary for possible seepage control measures:

- The permeability of the foundation soil or of particular strata in the foundation soil.
- The permeability is best measured by means of the in situ test.
- The shear strength of the foundation soil: To design the stability of the slopes of residue deposits it is necessary to have information on the shear strength of the foundation material.
- In many cases it may be necessary to consider both short term and long term stability conditions for a residue deposit.
- It is necessary to determine the erodability of the upper foundation soils in certain instances(24).

Open stormwater channels provide an economic and sometimes essential alternative to closed drains. The factor of safety against flooding of a channel is generally greater than that of a pipeline. A small water level rise in a channel increases the discharge capacity disproportionately whereas in the case of a pipe, the discharge capacity is only proportional to the square root of the head(25).

- *Penstock and Outfall Pipe* : Penstocks may be constructed of nesting reinforced concrete rings or as reinforced concrete brickwork towers. These penstocks function as a sharp crested shaft or "morning glory" spillway. Most of the time the penstock outfall pipe will carry only a small flow(24).

- *Under drainage* : Under drainage of the perimeter slopes of a residue deposit may be required to maintain or improve the stability of the slopes by drawing down the seepage surface. Under drainage systems are normally only required for the slopes of hydraulic residue deposits, but may in certain cases also be necessary to stabilise the slopes of residue dumps(24).

3.4 POLLUTION CONTROL

Mining has often resulted in pollution of surface water and groundwater resources. Therefore, polluted water must be dealt with in an environmentally acceptable manner. Keeping clean water clean, not only assists in reducing the risk of flooding the mine, but also results in less polluted water that has to be dealt with at possibly a considerable cost to the mine. Keeping clean water clean for different types of mining implies:

- For opencast mines minimise streamflow / surface runoff entering the pit
- For dolomite areas minimise surface water ingress
- For underground mines locate shafts above flood lines

Mining is cheaper where low stripping ratios occur as in valleys and this is where the water flows. It is thus often necessary to provide a water diversion system or water transfer scheme where an opencast mine is planned in a valley. A design recurrence interval is necessary when planning a water diversion that is commensurate with an acceptable risk of flooding. During a flood the water could cause severe damage to equipment, result in long delays and the water will probably be polluted by the time it is pumped out. This flow could have been attenuated in a stormwater control dam and diverted around the mine. Typical diversion works would consist of : dam + canal, dam + tunnel, dam + pipeline.

Sources of polluted water in an opencast mine are:

- Seepage from groundwater
- Seepage through spoils
- Rainwater falling directly into the mining area
- *Stormwater which is allowed to flow into the pit*
- Flow over poor or unrehabilitated areas

Some options in dealing with polluted water are:

- Evaporation
- Irrigation
- Underground storage
- Treatment
- Reuse on mine

Most of these options are very expensive thereby providing a good motivation to keep clean water clean and minimise the quality of dirty water(26).

Stormwater in contact with pollutants contained in or deposited on raw materials is potentially capable of carrying these materials into lakes and rivers. Surface and groundwater pollution may be controlled by the hydrologic and hydraulic measures required by the Water Act(24).

4. STORMWATER MANAGEMENT STRATEGIES

Ramps for open cast mines can be rehabilitated only at the end of the mine's life, when they become low-lying areas. They can serve as stormwater runoff control drains, directing runoff from rehabilitated areas into the final void. The whole of the new landscape could be higher in altitude than the surrounding unmined land. The groundwater table in the new landscape will eventually recover to a level dictated by the surrounding unmined rock types and topography(14).

By preventing air and rain water from getting to the pyrite by compacting coal discard in thin layers (the same prevention as for spontaneous combustion) the volume of storm water that percolates through the residue and is contaminated by pyrite oxidation is reduced(27).

Constructed reedbeds/wetlands have relatively low construction costs and operating costs, and may effectively be used to buffer peak storm run off.

Slimes dam storm water control can be subdivided into categories:

- *Top water control* - deposit tops should be shaped to force water away from the edges and to stop it spilling down the sides. Water must also be prevented from flowing from one side of the top to another. Rain falling onto the top of a dump should remain where it landed(14).
- *Side water control* - on steep-sided dumps, the amount of rain falling on the side can be controlled by vegetation although it will not manage stormwater runoff. Toe paddocks can also trap eroded material and polluted water.
- *Overland flow control* - stormwater flow must be diverted around the sides of deposits to stop it becoming contaminated or eroding the deposit's toe.
- *Seepage control* - most modern wastes are designed with underdrain systems to collect seepage and lead it to a collection point. On older dumps, a seepage interception trench excavated down to rock or an impermeable soil layer is required on the downslope side of the dump. This must also discharge safely to a collection point for recycling, evaporation or treatment(14).

The mine manager is responsible for the compilation of a plan that must depict all works constructed for the control of water on the surface of a mine or works and contain a list of prescribed details(18). The manager also has a number of specific duties to perform concerning mineral tailings and waste rock dumps, slimes dams and other sources of pollution, in respect of waterways, stormwater drains and dams, and with regard to domestic effluent.

Precipitation on a coal pile as well as the use of water for dust suppression and fire protection purposes results in surface run-off and leachate discharge from a coal storage pile. The need to protect the water quality often requires the construction of a water facility having sufficient capacity to handle the flow rates anticipated.

A new storm-sewage overflow was commissioned incorporating two proprietary Storm King overflow devices. The Storm King Overflow is a passive system that operates only when needed. Normal dry weather flow does not affect it. During a storm it comes into operation automatically, using the principle of dynamic separation to retain gross solids and floating matter in the foul sewer for full treatment at a downstream facility. Overflow is efficient in preventing the escape to the receiving waters of gross solids. The retention of suspended solid is another key area where Storm King Overflows excelled the tests. 35-60 per cent of the mass of biochemical oxygen demand (BOD) *entering Storm King Overflows during storm conditions were retained in the* throughflow. Effective reduction in what are commonly considered to be dissolved contaminants was 5 -45 per cent[31].

Stormwater management strategies include:

- Alter the activity
- Enclose the activity
- Cover the activity
- Segregate the activity
- Discharge stormwater into the process wastewater treatment system
- Discharge small, frequent storms into the public sanitary sewer
- Discharge small, high frequency storms into a dead-end sump
- Treat the stormwater with a stormwater treatment system[18].

Keeping pollutants out of stormwater is always preferential to removing them after the fact.

5. CONCLUSION

Adopting an integrated approach, engineering objectives, such as flood risk reduction, stormwater control and minimising disruption to production, can be simultaneously achieved with environmental objectives resulting in a minimum impact from the mine on the natural water resources[26].

To apply an effective stormwater management strategy, it is necessary for the mine to make a clear distinction between contaminated and clean land areas. Runoff from clean areas must be collected and routed around contaminated areas to be discharged to surface streams. Runoff from contaminated areas must be isolated and routed to pollution control dams from where it can either be discharged, reused or treated. Inflow to pollution control dams may contain structures to divert major flood flows around pollution control dams direct to surface streams. All pollution control dams and runoff diversion works and trenches must be designed on the basis of hydrological calculations for the catchment areas. Keeping clean and contaminated water separate will invariably result in reduced environmental impact and reduced water management cost.

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RESIDUE DEPOSITS

1. INTRODUCTION

Discard facilities, in most types of mining industries in the country, have a pollution potential. Due to some potentially hazardous substances, public concern and media interest is usually high. Structural failures like at Rustenburg and Merriespruit not only lead to loss of lives and adverse impact on the environment, but one such incident can cost the industry many millions of rands.

It therefore makes economical and environmental sense that these discard facilities are designed, operated, maintained, managed and rehabilitated in an environmentally acceptable way.

Residue includes tailings, the part of material that remains or results after processing to extract those constituents that are profitable to extract at the time; as well as waste rock which, not being ore itself, is rock removed from the mine in order to allow the extraction of ore.

2. CHARACTERISATION OF DISCARD FACILITIES

On coal mines, discard facilities range from "spoils" overburden, to lower grade coal. Dealing with residues from coal mining is an example of the complex issues relating to waste management. Part of the process is the characterisation of the discard material according to composition, susceptibility to spontaneous combustion, dust production, mineralogy and acid generation(1).

On gold mines, discard ranges from rock dumps to sand dumps to slimes dams. Textural analysis of 8 gold mine slimes dams (ranging in time of construction from 1888 to 1994), revealed little variation in grain size (around 0,1 mm), sand dumps were found to have a grain size of about 0,4 mm.(2) The grain size affects porosity and permeability, which in turn determines the seepage rates into ground water.

Cyanide, lead nitrate and zinc are used in the gold extraction process. All these elements are potentially toxic, and should be kept in a state that is not bio-available.

Gold slimes dams may contain radioactive radium (Ra), uranium (U) and other radionuclides. Radioactive waste is a material with an activity above an agreed level and that has no foreseeable use. In the activity of mining, wastes arise in slimes, rock dumps, sludges, soils and various products of uranium plants including hardware and buildings themselves. Licensing by the Council for Nuclear Safety (CNS) for the purpose of protecting public and mine *workers against nuclear hazard, has been initiated. Licensing includes laying* down of standards, assessment of risk, design of controls and implementation. Levels of 74 Bq/g or 3 700 Bq total activity used to be considered those at which attention is required, but a newer standard is 1 000 Bq/g ^{226}Ra or 1 000 Bq total activity. For ^{226}Ra in soil, 0,04 Bq/g is considered safe, 0,2 - 0,4 needs radon free design and typical tailings dam material is 0,6 - 4 Bq/g (waste rock 1,0 Bq/g). The CNS considers material with an activity of < 100 Bq/g as safe to dispose of in slimes, material 100 - 1 000 Bq/g could be diluted; and material > 1 000 Bq/g should be treated as radioactive waste (also see literature review on radioactivity)(3).

The 40 larger active South African gold and uranium mines annually dispose of approximately 120 000 000 tons of ore and 30 000 000 tons of waste rock. The smaller mines process roughly 1 100 000 tonnes of ore per annum, while 90 000 000 tons p.a. of sand dumps and slimes dams are reprocessed for gold, uranium and pyrite recovery.(4)

On larger mines, domestic waste disposal may also need maintenance along with the other industrial waste.

2.1 POTENTIAL ENVIRONMENTAL IMPACTS OF DISCARD MATERIALS

Coal overburden may have a lower pollution potential, as concentrations of contaminants are relatively low. Degradation of the soil does however occur over time and the sooner rehabilitation is initiated, the better.

Coal discard dumps containing lower grade coal contain chemicals that can contaminate surface and ground water, and further can cause air pollution due to dust particles and chemicals in smoke from spontaneous combustion.

Sulphur and associated sulphide and sulphate is the largest contributor to both air and water pollution from coal mines. These constituents contribute to salinity of the surface and ground water. Acid mine drainage may also mobilise metals in the water.

Gold slimes dams have a larger water retention capability than coal facilities, and have a larger pollution potential for aquifers they are situated on, though they may also affect surface run-off. Sulphate or chloride concentrations may increase salinity of water. Nitrate concentrations may be affected if lead nitrate gold extraction is used, this contributes to the nutrient loading of receiving water. In South African gold mining, 3 - 4 % of the gold bearing rock matrix consists of pyrite, which makes it an important environmental element in gold residue, as it is a source of ferrous iron and sulphate(5). Oxidation of pyrite in slimes dam material will lead to acid mine drainage in time. This process increases salinity as sulphuric (H_2SO_4) acid contains sulphate. High TDS concentrations can be expected and the low pH concentrations affect aquatic life directly. Furthermore, metals are brought into solution and become bio-available.

Methods of neutralisation of acid mine water with calcium carbonate (instead of lime) are proposed by some authors(6). A bench-scale biogenic sulphide and limestone treatment system has also been used to treat acidic metal contaminated water(7). As mentioned the low pH of acid mine drainage causes mobilisation of metals in water. Aspects of neutralisation are dealt with in detail in a separate review.

Metals like lead and zinc accumulate in biota (including humans) and may become toxic. Radioactive radiation from slimes containing uranium residue can be a health hazard and cyanide ingested - especially through contaminated ground water - can prove fatal. The mean lethal dosage for free cyanide for humans ranges from 50 - 200 mg, with death occurring within an hour. Toxicity and treatment of radioactivity and cyanide are dealt with in separate reviews.

The characteristics of residue having environmental effects may be summarised as follows:

- toxicity
- radioactivity and other health affects e.g. free asbestos content
- soluble solids content of residue and its acidity or alkalinity
- effects of oxidation and weathering on all the above mentioned
- possibility of spontaneous combustion of a residue
- possibility of soluble chemical content of residue may have an adverse effect on foundation material through ion-exchange.
- residue may contain substances toxic to plant or animal life
- possible weathering crust formation that may protect the residue from wind erosion
- content of chemicals that may precipitate out in the under drainage
- combustible or poisonous gasses from decomposition of substances in residue
- coal deposits contain carbon and pyrite, air flow induces spontaneous combustion and the subsequent heat increases acid generation
- Uranium tailings contain long-lived radionuclides and leaching of these radionuclides into water is generally regionally limited.

The following engineering or mechanical characteristics of residue should be obtained:

- Particle size analysis of residue; if it is possible to separate the residue into fine and coarse particles cheaply e.g. by cycloning, the coarser fraction may be used to build the outer slopes of the deposit.

- Permeability of the whole material and its coarse and fine fractions should be established. If there is stratification of the deposit, measurements of permeability should be carried out parallel and normal to the stratification.

- Shear strength characteristics of the residue should be established (in situ by means of a vane shear test), pressure meter test etc.

- Dispersion characteristics: if the chemistry of the pore water of certain soils is altered they become deflocculated or dispersed. In this state, soils are subject to easy erosion under the influence of percolating water. Foundation soil erosion in this manner may lead to dump failure.

- Hydraulically deposited residues may exist in a loose state. The loose state in combination with a high state of saturation may render the material liable to liquefaction.

Some of the following characteristics may change over time, due to physical or chemical weathering:

- Siltstone and mudstone can expand depending on moisture content and may rapidly weather to their original state as silty clays and clays

- Partly weathered igneous rock such as dolorite and kimberlite may also undergo rapid weathering from pervious granular materials to impervious clayey materials within a few years

- Coal residues are also subject to weathering. Pyrite oxidation is a major concern for water pollution in this respect.

- Gold mine residues are also subject to weathering of pyrite and shaly substances.

2.2 TYPES OF RESIDUE DEPOSITS

- Coal residue is subject to spontaneous combustion, and compaction and slurry mixing are methods of reducing air flow. Residue should be spread in layers not exceeding 300 mm in thickness that will compact to a thickness of 200 mm. Spreading of residue may be achieved by a moving bottom dump truck or by grader/dozer after dumping. If the dump is formed with side slopes less than about 18° to the horizontal (1 : 3), slopes may easier be compacted by placing a soil layer over the surface which may be lightly compacted.

- Residue dams retain or impound fine grained material which is deposited hydraulically. Cross valley impoundments, ring-dyke impoundments (complete ringdyke/side-hill dyke) and in-pit impoundments may be differentiated. Embankments of cross valley and ring-dyke impoundments are usually constructed of residue. Downstream method, centre-line and upstream methods of construction may be used during manual, mechanical or hydraulic placement of residue. Hydraulic -fill of nearly single sized run-of-mill residue may also be used for construction.

Every residue deposit can be considered to consist of three closely inter-related sub-systems:

i) Water control and balance system
ii) Seepage control system
iii) The slope, embankment or dyke system.

The water control system needs to meet the following needs (according to Government Notice No. R287);

- A system of storm water drains that diverts water around residue deposits, with provision for maximum precipitation in a 24 hour period with a frequency of once in 100 years and a freeboard of 0,5 metres.

- Rain water falling on the deposits, need to be retained in paddocks or on top of the deposit. A diversion drain and catchment dam is needed for polluted run-off.

- A water balance needs to be set up for each of the hydrological subsections.

- Design of penstock and outfall pipe, under drainage and sealing of under drainage, shear stability of dumps, outer walls of dumps, return water dams and coal slurry ponds are all outlined in the Chamber of Mines document(22) together with erosion prevention and seismic stability analysis.

The sub-aerial technique involves the systematic deposition of tailings in thin layers and allowing each layer to settle, drain and partially air dry prior to covering with a further layer. Underdrainage produces densities in excess of those achieved by sub-aqueous deposition and any air-drying serves to preconsolidate each layer with a resulting further increase in density. The low permeability of the tailings surface resulting from this deposition technique results in high runoff coefficients and, by decanting the runoff component to direct precipitation, a net evaporation condition can be achieved even in high rainfall areas(26).

2.3 OPERATION OF RESIDUE DUMPS

Dump subsidence or failure can have disastrous impacts on aquatic ecosystems so operation and maintenance practices should include precautionary measures.

Factors of residue dump operation that need to be brought into account are:

- method of deposition and distribution of residue
- rate of deposition (dam rise)
- amount of water stored on top of slimes dam and period stored

With careful and rational operating procedures it is possible to control the density, compressibility, permeability and shear strength in various zones of the residue dam.

There are two recognised classes of operation for residue dams: wet operation and semi-dry operation. Wet operation is usually adopted when residue is deposited within a dam which has been separately constructed of material different to the residue it contains. Semi-dry operation is the method generally adopted in South African climatic conditions where annual evaporation of free water exceeds rainfall. A ring discharge is used around the perimeter of the deposit, and the outer wall is raised by ploughing or manual shovel packing a dyke on either side of the wall area. The wall area is then further subdivided into paddocks by cross dykes. Each of the paddocks are filled in rotation with 100 to 150 mm of residue slurry which is generally at a water content of about 90 % by dry mass. The paddocks are allowed to dry for a period of two to three weeks before the next lift of material is placed. All building is carried out during daylight hours, and at night the residue is deposited directly into the body of the dam.

2.4 MONITORING OF RESIDUE DEPOSITS DURING OPERATION

The stability of the residue deposits depends on certain assumptions made during their design. The most important assumptions relate to the level and position of the seepage surface, the functioning of under drainage and shear strength with depth.

- Seepage surface height and position may be observed by means of stand pipe piezometers.
- Strength profile of material comprising a dam wall is best monitored by performing in situ vane shear or pressure meter tests.

Visual inspections, monitoring by survey and temperature measurements are further monitoring possibilities.

Stabilising fine coal refuse ponds for reclamation may be complicated by entrained water in stored refuse, that causes liquefaction as reclamation caps are placed. This may result in additional material requirements to those originally projected to stabilise the pond surface. Increased densities result in improved storage efficiency, less entrained moisture and a more stable refuse mass, thereby providing an improved working surface for reclamation and improved post-closure cover stability(25).

3. CHOICE AND EXPLORATION OF SUITABLE SITES FOR RESIDUE DEPOSITS

Major factors affecting locational analysis for residue deposit siting include; proximity to residential areas, available area, the distance from the reduction works or other plants that residue emanates from, topography of the site, presence of surface water in the form of streams, pans or vleis and the presence of the site to open-cast or underground workings, shafts or occupied areas. Additional factors that should be taken into account are; special or unique ecological properties, possible influence on agriculture, proximity to sand, rock and clay for under drainage, the geology of the area and groundwater aspects.

3.1 CLOSURE OF RESIDUE DEPOSITS

Specific features that should be noted for closure are: condition of seepage exit points and penstock outlets, geometry and material placed in embankments, penstock and diversion structures as well as the extent of erosion of embankments and abutment of river diversions.

Control of access to deposits and aftercare also need to be planned.

4. LEGISLATION CONCERNING DISCARD FACILITIES

The Department of Mineral and Energy Affairs produced a document: Aide Memoire for EMPR's in South Africa. According to legislation, mining companies and operations should produce Environmental Management Programme Reports. Monitoring and managing of discard facilities need to be addressed in these reports.

Of particular interest in the work of van Rooyen[9], among other topics, is the legislation around discard dumps, run-off and seepage from them.

The Environmental Conservation Act (Act 73 of 1989) Section 20 of part one states that only a disposal site for which a permit has been issued by the Minister of Water Affairs may be used for waste discard. The section is concerned with environmental degradation.

The Minerals Act (Act 50 of 1991) superseded the Mines and Works Act (Act 27 of 1956) Section 5.6.1 "If in the opinion of the inspector of mines the conditions of or the circumstances ...... of dangerous slimes dams, waste dumps .... whether abandoned or being worked, are dangerous to the life or health of persons, property of public traffic he may order that it be safeguarded to his satisfaction by the owner or manager of the mine or works."

Section 5.10 "To prevent the dissemination of any form of pollution such as dust, sand ...... of any dump, the manager shall cover the dump with soil or sludge or otherwise deal with it in a manner satisfactory to the inspectors of mines."

The Water Act (Act 54 of 1956) Section 26 (c) and (d) Regulation 5.1 of government Notice No R 287 states: "The manager of a mine ... shall cause a plan depicting all works constructed for the control of water on the surface of a mine .... Such works may include: (3) mineral, tailings and waste-rock dumps and slimes dams with the proposed annual extensions for the following five years.

Regulation 7 of the above act: "The manager of a mine ... shall make adequate provision to the satisfaction of The Secretary to prevent as is reasonably practicable, run-off from eroding slimes dams and mineral, tailings and waste rock dumps due to rain."

Regulation 11.1 : ".... all slimes dams and minerals, tailings and waste-rock dumps to be .... constructed or modified that all rain water precipitated there on, is retained there on..."

Regulation 11.3 : " French drains and ditches to catch seepage water and carry it to evaporation dams .... shall be constructed at all tailings dumps wherever seepage occurs."

Government notices No R 553 (1962) and No R 909 (1971) to prevent water contamination were released, and effluent standards were set up in the documents; development around these standards has frequently been noted [9].

In the setting up of management guidelines, different options could be suggested for specific circumstances and financial situations could be taken into account.

5. ENVIRONMENTAL MANAGEMENT AND REHABILITATION

EMPR's that need to be completed by mines, for government authorities, include discard management and pollution control. It must contain clear schedules of planned rehabilitation work, and negotiations should be carried out with representatives of the government departments, especially the Department of Water Affairs. The EMPR is a dynamic document that is owned by the mine and the authorities. Every mine should have specific persons responsible for the EMPR[10].

Sections of the programme from the EMPR's may be subjected to auditing, and has also been suggested to be made public[13].

Misperceptions by media include, "hills" (dunes) to be mined at St.Lucia and "The Environmental Impact Assessment done by Richards Bay Minerals is the first, and until now the main concern of the government has been the protection of soil".

Statements like these in newspapers, can necessitate large "education" campaigns in order give the public a truly realistic view. The availability of *information to the public is also encouraged by Gunderson, "before they (the* public) are misinformed"[14].

The development of an Environmental Management Programme for the Spitzkop Colliery water aspects, included a hydrological survey, as it was alleged that the mine was a major polluter of a dam that supplies a local town with water for domestic use. This showed that the water quality of the dam actually improved on those occasions that there was major run-off from the mine to the dam.(15)

About 50% of the total water intake by SA gold mines consists of abstracted river water. Low quality mine water is eventually used for dust suppression, washing and milling pyrite flotation and disposal of tailings. As tailings and mine water eventually end up on slimes dams, which are not closed systems, it makes sense to reduce the amount of contaminants wherever possible - right from where water is extracted through to disposal(16).

Rehabilitation of coal overburden has proved successful, to an extent where rehabilitated areas can be used for agriculture again (grazing for livestock). For coal discard dumps a method of mixing slurry and coarser coal has been devised to reduce air flow through the dumps, thereby reducing spontaneous combustion.

For gold slimes dams, revegetation with grass species have proved successful, but costs are considerable. There are essentially 4 traditional techniques for vegetating residue deposits:

The dry land method by hand, dry land by machinery, hydroseeding and leaching. Gold tailings are low in nutrients, become acidic, and erode easily. Dry land vegetation by hand is the reed paddock method for good rainfall areas where pH is not too low in the tailings. Dry land vegetation by machinery applies when the surface is fairly level, and agricultural equipment can by used. "Potato" ridges are constructed and lime is applied. Hydro seed consists of blowing a water, seed, fertiliser mixture onto steep side slopes. High mulch and lime rates are difficult to achieve. Leaching is used in the presence of toxins and acidity, but a great deal of water is needed and lime mulch and phosphates are added(17).

A new method of using trees, together with grass (any of the above methods) and wetland species, can reduce seepage more effectively, remove potentially toxic metals from the slimes dams and ground water, and produce timber.(2) The use of wetland systems in primary water treatment (18) and metal accumulation(19) from water is discussed and WRC funded research in treating mine water passively in this manner will start in 1995.

5.1 . ALTERNATIVE RECLAMATION SCENARIOS

Candidate reclaimed configuration alternatives should be developed to assess plausible post-mining land forms. The general configuration of the tailing facility surface assumes minimal surface grading following cessation of tailing deposition. Key to this assumption is the efficient consolidation of stored tailings. Each alternative introduced below establishes a method for improving tailing consolidation in order to provide a suitable working surface for reclamation equipment.

In addition to stabilising stored tailings, reclamation alternatives should consider the ability to mitigate potential for acid-generating conditions and provide for adequate stormwater management and control with minimal impact to milling operations. To this end, three plausible "generic" alternatives were developed and analysed:

- Passive dewatering
- Blanket dewatering
- Selective dewatering[23].

Establishing preferential pathways is necessary to achieve substantial drainage from a tailing mass. Implementation of the managed deposition system will establish a more consolidated tailing bench and will also reduce the phreatic surface within the stored tailing. Minimising operating water levels and establishing a consolidated beach serves two key purposes in serving reclamation objectives:

- It creates a more consolidated "crust" or cap above the less consolidated stored tailing. This results in a layer of material which may be suitably stable for reclamation purposes.

- The consolidated layer provides a surcharge loading on the less consolidated stored tailings. The surcharge then pressurises the pore water stored in the less consolidated material below which will cause pore water to diffuse along preferential pathways to relieve pressure. Typically, because the beach is formed in "wedges" with the thickest portion being nearest, and the slurry distribution system thinning to the water pond area, the preferential pathway for pore water relief should ideally be in the direction of the water reclaim pond.

The degree to which preferential pathways are established in the stored tailings to relieve pore water pressures, establishes the basis for the reclamation scenarios evaluated. In general, the cost to implement each alternative is commensurate with the level of supplemental pathways established.

5.2 PASSIVE DEWATERING

The passive dewatering alternative constitutes a "no action" approach in terms of introduced preferential pathways. It assumes that the consolidated tailing beach, which is established by managed slurry deposition, will be suitably consolidated to support reclamation loads on the beach surface. Because the materials adjacent to a localised decant pond will be saturated during operations and the consolidated beach thins toward the pond area resulting in unsuitable conditions for stable cover material placement, this option inherently assumes that the post-mining land use should include wetland and seasonally wet areas within the reclaimed facility to take advantage of this condition: i.e., don't try to fight it.

Passive dewatering assumes that reclamation equipment can work on the surface of the consolidated tailing beach to place soil covers over the stored tailing. Acid-generation potential for stored tailing is typically greatest closest to the discharge source, diminishing with distance from the source to often non-acid-generating conditions. This is because pyrite, the primary source of reactive sulphur for acid production, is almost twice as heavy as a quartz bearing tailing and will tend to deposit closer to the source as long as the particle size distribution of the two components is similar. The area where covers can be effectively placed will be those which are closest to the deposition source. This allows for mitigation of some, if not most of the potential acid drainage while utilising the higher and better land use of wetlands and seasonally wet areas in the non-acid-producing portion of the facility.

5.3 BLANKET DEWATERING

Several alternatives were evaluated, to determine the most suitable introduced preferential pathway scenario for relieving pore pressures in the stored tailing. A brief description of each follows.

5.3.1 Constructed Drains and Sumps

Constructing drains and sumps in the in-place tailing were considered. This method would require wrapping drain gravels in geotextile or other pervious fabric and placing it into the in-situ tailing. Drains would be channelled to a sump where dewatering would occur by pumping. This option was eliminated from further evaluation based on the cost of installation and pumping requirements.

5.3.2 Drilled Sumps

This alternative considers drilling on-angle through the embankment to establish drainage sumps. Drill casing would require a filter-fabric wrap to minimise potential for slurry intake to the sump. Dewatering of sumps would be by pumping. This option was eliminated from further consideration because of the localised drainage area it provides, cost, and pumping requirements. Horizontal drains may be possible to provide passive dewatering, but the logistics of drilling into unconsolidated slurry negated serious consideration.

5.3.3 Geonet or Tensar Intermediate Layer

Consideration was given for placing a blanket of drainage material or high tensile strength material over the surface of the stored tailing. This option considered placing either a layer of Geonet sandwiched between two geotextile layers or a layer of Tensar (Tensar Corporation, Morrow, GA) prior to implementing the managed slurry deposition.

The Geonet option establishes a preferential pathway between the unconsolidated tailing and the managed beach and relies on the surcharge load to force pore water upward and into the drainage layer. Actual experience at some operations supports this concept in that upward flow of liquefied tailing is the normal mode of pressure relief under loading.

The Tensar option does not establish a preferential flow path; it essentially "suspends the load" of the consolidated layer. The material strength of the managed tailing beach, coupled with the Tensar support to support the load should result in a stable tailing surface over the long term.

An intermediate layer of Geonet/Geotextile composite or Tensar appears to be a viable method to stabilise tailing surfaces for reclamation. The material cost is approximately $ 0,60/sq. ft over the surface area of a tailing disposal basin. For a 100 acre site, this unit cost corresponds to an approximate material cost of 2,6 million dollars. These options are considered interchangeably or in combination with the wick drains subsequently described for comparing and ranking of the candidate alternatives.

5.3.4 Wick drains

This option considers placing vertical wick drains into the unconsolidated tailing. Wick drains were developed to relieve pore water pressures in saturated soils in order to establish a stable surface for construction. Wick drains have been used in the construction industry for dewatering road and building foundations prior to construction.

Wick drains are designed on the premise that surcharging a load on the saturated material will force the pore water into the vertical drain for pressure relief. The weight of the surcharge forces the water in the drain to flow "uphill" and out of the drain at the surface. In addition to their successful use in the construction industry, Knight Piesold recently installed wick drains into gold mine tailing in Montana. Flow response from the emplaced wick drains was rapid without external surcharge. The weight of the in-situ tailing was sufficient to begin dewatering the lower portions of the slurry.

Design criteria for wick drain spacing requirements is assumption-dependent. For the purposes of evaluation, Knight Piesold assumed that tailing stabilisation and consolidation must occur within one year, and the optimum consolidation rate was 90 percent. Based on these assumptions and a nominal 80-foot wick length, the influence of the vertical drains would be about 5,2 feet. Therefore, assuming 10-foot spacing over a 100 acre "hypothetical" tailing basin surface area at an installed cost of approximately $0,75/ft, the cost of wick drains is about $2,6 million (the same as the Geonet/Tensar option).

5.4 SELECTIVE DEWATERING

This option considers a combination of the Passive Dewatering Alternative and the Blanket Dewatering Alternatives previously described. It takes advantage of the capital cost saving of passive dewatering, but accounts for limited active dewatering using wick drains or Geonet in localised areas. Unlike the drain installation plans, drainage provisions under this scenario would be installed at the time of reclamation rather than prior to the implementation of the managed deposition system. This provides for a further cost savings associated with delayed expenditures.

It is possible that improved stabilisation techniques will be needed over some portion of the facility. This option allows an operation to take advantage of the improved consolidation associated with managed deposition prior to making the decision to install drainage provisions. This time element allows for the beach to surcharge the saturated tailing. Pore water is expected to diffuse toward the decant pond under this loading. It may be that the surcharge loading would be sufficient to consolidate the tailing to a stable condition(23).

Fluid collection and removal in a fully-drained mineral waste disposal facility is divided into three main categories: underdrainage, supernatant or surface water, and internal embankment seepage. Underdrainage and supernatant fluid account for more than 95 % of the total fluid collected within the facility. Based on the results of laboratory studies as well as field measurements of flows, approximately 50 to 70 % of the total slurry water content is liberated as supernatant runoff as the slurry undergoes natural settling. Underdrainage may contribute an additional 10 to 20 %, with the balance either permanently locked into the soil structure or lost by evaporation or subsurface seepage. In areas where water conservation is a key issue, water recovery by underdrainage alone offers a cost savings over the facility lifetime greatly in excess of the initial construction cost (24).

6. MODELLING FOR MINE CLOSURE

After all mining activities have been stopped, discard facilities may still be a pollution problem and closure certificates from authorities may not be granted. Remediation by this time will be more expensive than management of discard during production.

During the initiation and operational stages of mining, the pollution situation at closure needs to be anticipated for effective pollution control and management. Modelling of hydrochemistry can play an important role in pollution control and discard management. Not only can anticipated pollution be restrained, but a good forecast of what the situation will be like in years to come, can prevent over spending on current measures taken.

"By considering closure during the design phase of a tailings dam, cost effective measures for mitigating long-term environmental impacts can *readily be incorporated*" (20). Considerations, by these authors, for the closure of tailings dams, include; stabilisation, control of air and water pollution and erosion at a low maintenance cost, in order to obtain a closure certificate.

Reclaiming of gold and uranium from old slimes dams has led to relocation of tons of tailings from older slimes dams to newer ones with under drainage - which means less ground water seepage. Considerable amounts of slime is now also being used for backfilling (21), which reduces the amount of slimes that has to go to slimes dams. Usually blasting with water cannons is used, but dump recovery by bucket wheel excavator and belt conveyor has also proved useful and less water is used (22).

Allocating ground water pollution to specific discard facilities is a complicated problem, especially for neighbouring mines after closure. Radiogenic strontium and lead in water have isotope signatures which are determined by the geology of the source material. Lead is associated with sulphide minerals and exhibits varying isotopic signature depending on the age and type of sulphide mineralization. Lead isotope ratios may be utilised to estimate relative proportions of pollutant mixing in water systems contaminated by two different sources(31). This can be applied to apportioning contributions of acid mine drainage from different discard dumps in a area. Examples in gold and coal field areas exist.

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UNDERGROUND WATER MANAGEMENT

1. INTRODUCTION

The mining industry uses water for a wide variety of purposes in underground operations. Water is used underground in drilling operations, dust suppression, environmental cooling, condenser circuits on refrigeration plants and recently as an energy source in hydropower and as a transport medium in backfilling. In addition, potable water is supplied underground for drinking purposes. Many mines also produce considerable amounts of water through fissures.

Considering the rapid increase in mining costs and proliferation of safety laws, the need for greater attention to dewatering, particularly in the feasibility and closure stages, becomes apparent(3).

The gold mining industry in South Africa consumes and circulates an estimated 73 800 ℓ/s (6 400 Mℓ/day). Approximately 63 600 ℓ/s (5 500 Mℓ/day) of this water is circulated in closed loops. The bulk of this water (34 700 ℓ/s) is circulated as condenser water for evaporator cooling circuits, refrigeration plants, compressors, etc., while 9 800 ℓ/s of water is circulated to bulk air coolers, to cool the air underground. A further 12 100 ℓ/s of water is circulated for mining purposes, of which 4 800 ℓ/s is chilled water, which performs a supplementary cooling function in the stopes. A full 78 % of the water in circulation is associated with mine cooling(1).

2. UNDERGROUND WATER CIRCUIT POLLUTION PROBLEMS

Because many of the underground water management problems and strategies are discussed in individual literature reviews (acid mine drainage, neutralisation, chemical contamination, microbiological contamination, disinfection, desalination, filtration and settling, boilers and cooling, radioactivity, cyanide) these subjects will only briefly be discussed, and the focus will continue on fissure water, groundwater influx and preventing flooding of the work face.

Water quality management problems in underground water circuits are primarily manifested through high reticulation system costs. The prime motivation in addressing these problems is therefore to reduce the costs associated with the damaging effects of erosion, corrosion, fouling and scaling. Health and safety of workers also need to be considered(1).

Due to the complexity, and the large amount of variables that contribute to underground water circuit problems, it makes sense to model the expected water quality in mine service water circuits. In this way, a pro-active water management plan can be compiled, and problems can be foreseen and perhaps prevented. The mechanisms and chemical processes underlying mine service water salination, conditioning and purification has been investigated for a number of gold mines in South Africa.

Based on the results of these processes, unit process models have been developed to form building blocks for a system model for simulating water quality in underground mine service water circuits. The unit process models developed include a stope leaching, a neutralisation and a desalination model and were used to develop a mine service water circuit simulation program, AQUA-Q. Total dissolved solids, alkalinity, calcium and sulphate concentrations as well as pH are simulated at specific points in the system as functions of time. Case studies, have proved the applicability of the model for design of underground mine service water circuits, and the development of operational strategies for existing water circuits(11). More detail on AQUA-Q and underground water management strategies is provided by Pulles (12).

The use of computer software (GIS) in modelling and management of mine water circuits, is also developing rapidly and a review on Information Technology (Geographic Information Systems & Database management) is included separately.

The underground water problems can be divided into the following:

2.1 DISSOLVED SALTS

Acid mine drainage (which is discussed in detail in another literature review) increases salinity of most mine waters.

It has been shown that on average, mine service water will experience an increase of 20 per cent in the dissolved salts due to leaching in the stopes. Ammonia and nitrates enter the water circuits from explosives, and calcium or sodium levels are increased by addition of neutralising agents.

Cooling water circuits (bulk air coolers, spray chambers) experience an increase in dissolved salts due to evaporative losses and adsorption of airborne salts.

Water quality problems resulting from increased dissolved salts depend on the nature of the salts and the subsequent usage of the water. Salt increases may lead to corrosion problems, depending on the materials used in the particular water reticulation system.

Corrosion is a complex process and may be a function of many water quality parameters, such as dissolved oxygen content, pH, concentration of sulphate and chloride ions, TDS content and the degree of biological activity.

Corrosion could lead to some practical problems like increased friction loss in pipes, increases in suspended solids may occur - increasing erosion (abrasion) and blockage of equipment, uniform thinning or localised penetration caused by pitting or crevice corrosion can lead to pipe and fitting failure, high calcium contents lead to scaling of pipes and heat exchanger tubes(1).

2.2 ACIDITY

Acidic water leads to corrosion, while high alkalinity could lead to scaling. High acidity will also reduce settling and flocculation ability.

2.3 SUSPENDED SOLIDS

Spent mine water reaching settlers may contain between 100 and 40 000 mg/ℓ of suspended solids. Problems that this may create include; silting up of clear water dams, erosive wear on high pressure multistage centrifugal pumps, fouling and loss of efficiency in heat exchanger tubes in refrigeration plants and ineffective water disinfection - leading to health risks(1).

2.4 MICROBIOLOGICAL QUALITY

Underground spent water is contaminated with human faeces and urine. This water is often put through refrigeration plants and reused as mine service water. Workers do, however, sometimes prefer to drink this water instead of the potable water provided. If the chilled water is not adequately disinfected, the drinking thereof may lead to spread of diseases such as cholera, typhoid or dysentery(1). Apart from the health threats of these diseases, an epidemic among workers could affect productivity.

2.5 RADIOACTIVITY

South African gold mines contain ore with between 0,015 and 0,05 per cent uranium. The radioactive material contaminates spent mine water, and although settlers remove most of the radionuclides, some is retained and could be ingested by workers when the water is consumed(1).

2.6 BACKFILLING AND TRACKLESS MINING

Water draining from backfill paddocks, may be extremely polluted and may contain high concentrations of suspended solids and cyanide. The cyanide becomes a problem when the backfill water enters the mine service water circuit. If the cyanide concentration is high enough it can also leach gold from fines in the drains, which will then be lost with the discharged waste. Ferrous sulphate is often used to immobilise cyanide in the backfill.

Use of diesel LHDs in mining results in fuel and oil spills, which are a problem in the settlers(1).

3. ADDITIONAL CONSIDERATIONS

Saul (1970) describes the nature of mine drainage problems in terms of three categories; colliery closures, water on roadways for working mines and water on working faces.

In coal fields where many mines have been closed, the cost of maintaining increasing numbers of pumping stations at closed mines, or of pumping much increased volumes at greater depths in working mines, is a heavy financial burden on the reduced number of mines left to carry it. The same financial implications for gold mines may also occur.

The relatively complex hydrogeology of karst aquifers is often detrimental to mine production. Through experience, China has developed several different methods to control such mine water inflows. Selection of a method should be directed towards controlling the effects of prominent hydrogeological characteristics in different karst types. *For Karsts, surface water flows, water* cut-off ditches and subsequent diversion trenches are used. For ground water flows in semi-confined aquifers, drainage by deep wells and combinations of special water cut-off drifts and predrainage by galleries are used. In addition, *in confined aquifers such techniques as multiple depressurising wells (that* reduce fissure water entering workings) and water sealing wall operations (that seal of old work faces to prevent water influx) are used.

Multiple well systems used in one case study coal mine, prevented flooding for 18 years, increased mine production capacity six fold, reduced timber consumption from 550 m^3 to 230 m^3 / 10 000 tons of coal mined. The pumping costs however rose to 20 % of total cost of raw coal produced. This technique *is therefore combined with sealing wall operations and cut-off drifts at the* surface. Sealing wall operations are justified by huge electrical as well as pumping cost reductions. Furthermore, there is no impact (draw down) on the regional water table, unlike dewatering practices(5).

3.1 UNDERGROUND ARTIFICIAL DAMS

As part of underground water circuits, mines often have underground reservoirs for temporary storing of water. Blasting and seismic activity may affect the structure of such dams over time, and failure of such a dam could have disastrous results underground.

An approach based on "thick base theory" for underground impoundments is supported by many authors. The influence of blasting on a dam design, and a method to estimate the dynamic stresses due to blasting are suggested. Gupta et al (1987) suggest only arch shape dams of cement concrete. If the thickness of a dam exceeds 3 - 4 meters a multistage dam is suggested to prevent structural failure(7).

3.2 UNDERGROUND FISSURE INFLOW

As mentioned, most South African mines have water entering the work areas, from the surrounding geological strata. This water is usually subjected to pollution underground, but can be more of a threat to the ore extraction process and personnel, if fissures appear and the work area is flooded.

Floods during shaft sinking and mining operations, should therefore be included in hazard or disaster management plans.

Several examples of fissure flooding are discussed in literature (e.g. Blyvooruitzicht 1937 & Free State Geduld 1957). The techniques used at these mines were also applied at Western Deep Levels (1982 - 1983)(2).

Basic laws regarding groundwater flow have been known since the 1800's. Darcy, in 1856, developed what has become known as Darcy's Law from an empirical experiment on flow through sand. The law states that under laminar flow conditions, the rate of water flow through a porous medium is directly proportional to a constant, times the head loss over the length of flow and the area through which the water flows. The constant has been termed the coefficient of permeability, or more recently, the hydraulic conductivity.

A method for calculating the hydraulic conductivity, using data from a pumping well and at least one observation well, has been devised. The equation is known as the Theim equilibrium formula, and is still in extensive use today in solving ground water flow problems. The concepts have been developed over time and are currently in common use. The formula created is used for estimating the storage capacity and permeability of aquifer systems.

i) Darcy's law: $Q = PIA$

Q = discharge
I = hydraulic gradient
A = area through which discharge occurs
P = permeability (K) in m/sec

ii) Theim equation: $T = \frac{QLn(r2/r1)}{2\pi(s1-s2)}$

T = coefficient of transmissibility
(T = P times aquifer thickness)
Q = discharge
r1 & r2 = distances from pumped well to first and
second observation wells
s1 & s2 = down draw in observation wells

iii) Theis equation: $s = \frac{114.6}{T} \frac{e^{u}}{u}$ du

T,Q,r and s same as Theim
$u = 1.87r^2S/Tt$
t = time since pumping started
S = storage coefficient

These formulae are based on the principles of wells and the analogies of shafts/mines to wells, are used for dewatering assumptions. Dewatering cones can be modelled with the equations as well as the time required to lower the water table a required amount(3).

It must be kept in mind that in relatively impermeable rocks, single fractures can dominate the local hydrogeology. Faults have been shown to act as either barriers against or conduits for the movement of groundwater. At the investigation stage it is difficult to determine their effect on the hydrology, due to the low level of core recovery which usually occurs in fault affected ground (9).

Water on the working faces of coal mines causes slurry formation, which can cause operating problems during mechanised ore extraction. Algae have been used as an indicator of an external surface water source influx. Algae can be an effective tracer for determination of surface water sources(10).

In the coal fields of Northumberland and Durham workings under the sea, major dewatering problems exist, with some similarity to the conditions in the west Witwatersrand gold mines. Water influx to the work face can be reduced in these undersea mines by dewatering boreholes above the work face(4). In South Africa, gold mining areas are geologically in proximity to the karstic Transvaal dolomites, and have given rise to major dewatering problems. With *Mean Annual Precipitation at 750 mm, a 10 % recharge is estimated.* It has been shown that surface wells to a depth of up to 150m can withdraw significant quantities of water and reduce the amount that has to be pumped from considerable depths, with significant costs savings.

"Drilling technology is not a fast moving industry, but much smaller pumps are the order, and it is feasible to use boreholes instead of shafts to contain pumps, or else to use the holes to convey discharge waters to the surface from pumps located in drives or stopes" (13). Mineral holes drilled to assess ore bodies, could also be used to predict dewatering costs where possible.

At Orapa and Letlhakane diamond mines in Botswana, ground water influx increases the cost of drilling, blasting and instability of slopes. Deep wells, drainage galleries, toe drains and the control of surface water is applied to limit water inflow to the work face(8).

4. MODELLING FISSURE WATER

Aquifer constants may be calculated from pump-out and pump-in results. Groundwater data needed to determine flow rates and pumping capacities during longwall pumping can be obtained by various pump-out or pump-in tests.

Pump-out tests, water may be air lifted from aquifers for periods from 6 to 27 hours and flow rates recorded at v-notch weirs. Draw down can be monitored in observation boreholes located at different radii from the production holes. The Jacob and Theis methods may be used for analysis of draw down and recovery data. As the Theis method has already been discussed in the previous section, only the Jacob method will briefly be discussed.

According to the Jacob method, transmissivity (T) expressed as m^2/day is calculated as follows;

$$T = \frac{1.584 \times 10^4 Q1}{Dh}$$

where:
Q = flow rate
Δh = change in draw down per logarithmic cycle of time
W(u) = well function (exponential integral) obtained from type curve
$h\text{-}h_o$ = draw down in metres at time t

while for "S" the storage coefficient;

$$S = \frac{1.559 \times 10^{-3} Tt_o}{R^2}$$

t_o = time intercept in minutes to zero downdraw
R = radius from production borehole to observation hole in metres
u = variable from type curve

for k (permeability in m/sec);

$$K = \frac{T}{b \times 86400}$$

b = aquifer thickness

The pump-in open-end tests are normally done in unconsolidated or weathered material, where the hole is fully cased and the amount of water accepted by the ground through the open bottom is recorded.

The permeability (K = permeability in m/sec) for an open-end test is obtained through the equation:

$$K = \frac{Q}{5.5rH}$$

r = internal radius of casing or hole tested in metres
H = corrected differential head of water in metres.

Mine inflow rates are dependent on aquifer permeability and height of the aquifer above the mined seam. Aquifers with permeability > 10^{-5} m/sec can be successfully predrained with a well pit system (17).

Hydraulic parameters for inflow calculations are obtained from pump-out tests. Darcy's equation (as discussed in section 3.2.) may be used to relate flow rates from aquifers of different permeabilities once caving characteristics have been established.

Water temperature may be used as a tracer of ground water flow and its origin. The water temperature recorded during some studies provided an increase of 1°C each 30,3 metres of depth with a correlation of r = 0,97 (27). Similar thermal gradients may be determined by borehole logs.

The accurate prediction both of the water ingress into sub-surface excavations and the nearby groundwater pressures is of fundamental importance to the control of the stability and flooding of mines constructed below the water table. Predictions are usually based on laboratory and in *situ permeability determinations together with observations of groundwater* conditions in boreholes(19).

Although established theoretical methods provide a basis for prediction, the actual ingress depends on many factors. The exploration of the geological controls on groundwater conditions, and the available methods of prediction of both groundwater flow and pressure could be useful tools in the mining industry(19).

Measures to bring the underground water under control may be based on the principle of comprehensive ways and on the systems engineering method. Macroscopically, it is a reasonable countermeasure to retain surface water in the upstream section of a karst/dolomitic water system and to strengthen infiltration in the midstream section and to combine exploitation and utilisation of the groundwater with water protection and mine dewatering in the area in which mines are distributed (18).

Computer modelling of the ground water data has also been undertaken *adopting representative values for the hydraulic parameters and various* aquifer geometrics. Both analytical and two-dimensional finite difference models may be set up and the model configuration and parameters established by calibrating model predictions against historical data. The two-dimensional finite difference model might not always be able to adequately reproduce historical groundwater trends at a site. On the other hand, the more simplified analytical model may successfully be calibrated and consequently used to simulate the first six month dewatering requirements of an open pit(18).

In some case studies, the results of numerical modelling of dewatering requirements indicated that up to 82 operating bores and more than 100 bores in total would be required at any one time to intercept groundwater throughflow and to dewater ahead of mining. The boreholes will often need to be of large diameter to maximise yield when water-levels are lowered to near the base of the aquifer, but even so, some secondary in-pit means will be required to complete dewatering. Air-injection reverse circulation drilling is considered to be the most suitable means of installing large diameter bores in unconsolidated sediments. A large cone of depression could form around a dewatering borefield.

A conceptual model of equivalent fracture flow may be used rather than an equivalent porous medium model. One advantage of this method is that transmissivity of individual fractures is estimated and hence possible sudden bursts of water can be predicted with more confidence.

A second advantage is that the effect on flow caused by widening and narrowing of fractures around the perimeter of projected mining operations can be assessed more realistically in the future. Predictions are made both of flow from individual fractures as a function of depth and of average flow as a function of rate of a face advance. Three types of flow are distinguished, namely pressure-induced flow, gravity flow and long-term flow in response to regional recharge.

Most groundwater flow models are not sufficiently detailed to allow an explicit representation of all dynamical scales. Instead, most models are constructed at such a coarse scale of resolution that unresolved subgrid variability often exists. A local Darcy's law is a universally valid resolved scale model if the resolved and subgrid scales of the hydraulic conductivity are separated in scale by a so called spectral gap. If the hydraulic conductivity possesses many scales of variability, then a more general non-local Darcy law is a more appropriate model structure. When the non-local Darcy law is appropriate, numerical experiments suggest that errors using a local Darcy's law with effective parameters are most significant at the smallest resolving scale of the model, and are minimal at scales between 8 - 16 times the resolving scale(23).

Numerical groundwater models are playing an increasing role in the analysis, understanding and prediction of fissure and groundwater behaviour. Essentially, a numerical model is a computer program which solves the set of equations which describe the physical system under examination. Modelling techniques can be used to examine highly complicated systems which could not be sensibly evaluated by manual calculation methods. There are numerous potential applications within the mining industry, from initial feasibility studies, through design and optimisation of dewatering networks to assessment of contamination problems arising from spoil heaps and analysis of post quarrying landfill activities (22).

The quantification of underground water flow into open pits, is currently the most common application of modelling techniques in the extraction industries. Models can actually help in minimising the amount of data that needs to be collected, often resulting in considerable cost savings. Preliminary models can be established using available data: the models can then be used to examine a range of development scenarios using different values to assess the sensitivity of the problem to these different distributions. This allows for the enhancement of variables and specific areas that need to be focused on. Further data collection can then be concentrated on these aspects identified. Models can be used to quantify flow toward mining operations through its development. Possible dewatering schemes can be experimented with in the model to establish the optimum location of dewatering wells and optimum pumping strategies, to ensure that dewatering always keeps ahead of excavation. Known unit costs can easily be applied to each of these possible options to minimise economic impact.

A typical study which was involved in such operations was carried out at an open pit gold mine in Nevada, USA(22). Preliminary drilling, geological, geophysical and hydrogeological surveys produced data which were assimilated into a 3D model which was used for predictive simulation of dewatering and examination of the effects of abstraction from different aquifers. The model predicted that pumped discharge would increase from 60 Mℓ/day to 100 Mℓ/day over a 8 year period. Pit abstraction water was discharged several kilometres from the pit, however, the investigation demonstrated that around one third of the water pumped was recirculated, so alternative disposal methods were put in place.

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WATER RECLAMATION

1. INTRODUCTION

The average annual rainfall of South Africa is approximately 480 mm which is almost half the world average of 860 mm. Compounding this problem is the bad time and spatial distribution of this precipitation. For the major portion of the country, rainfall events are characteristically seasonal, short and intense. The proliferation of dams and storage facilities alleviate this problem to a certain extent. A number of inter-catchment transfer schemes have also been developed to redistribute the water resources more evenly between the eastern and western areas of the country. South Africa is heavily reliant on surface water resources(1) and with an evaporation rate varying between 1100 mm in the east and 3000 mm in the west, the potential water shortage problem is evident.

In addition to the limited water supply, the domestic, industrial and agricultural demand is increasing at a rate which implies the total fresh water demand will exceed supply in approximately 25-30 years time(1).

There is thus an increasing awareness and motivation, among the larger users, for optimisation of water utilisation. The strategies employed involve recycling and reuse of existing water, minimisation of water losses such as evaporation and seepage and a reallocation of water resources to users, taking account of the user's fitness-for-use criteria. However, the awareness of the scope of the problem needs to be better conveyed to the general public as there is an apparent resistance by domestic users to water reuse and recycling, even in arid regions(2),(3).

2. OBJECTIVES

The objectives of water use optimisation, by way of reclamation, are motivated by environmental consciousness as well as by economic considerations. The reduction of effluent has both an environmental and economic basis since failure to comply with increasingly strict legislation results in financial penalties.

2.1 WATER CONSERVATION

There is a growing awareness of the current and increasing shortfall between existing water supply and the ever increasing demand. Satisfying this excess demand will involve substantial cost(4) and it is thus imperative that water conservation strategies are developed. Conservation of water can be considered in terms of both the volume available and the quality thereof. The conservation of water quality has recently received the greatest attention through pollution reduction measures but the minimisation of unnecessary water loss needs to be addressed urgently.

2.2 WATER COST REDUCTION

The motivation to conserve water as described above may be lacking in countries where inadequate education has resulted in a poorly developed environmental consciousness. However, the cost of water and water treatment are tangible concepts that are easily understood by domestic, agricultural and industrial users. The objective of reducing costs in the face of increasing competition is likely to remain the most important motivation for optimising water utilisation.

The cost reduction may be through reduced consumption of potable water which is usually paid for or through a reduction in water treatment costs. Consumption of good quality water may be reduced by substitution of a worse quality water to a particular user or through a number of water saving strategies. Water treatment costs may also be reduced by an analysis of the user fitness-for-use criteria and downgrading the treatment accordingly. These options need to be evaluated as part of an integrated water management plan as described briefly below and in the literature review dealing with water and salt balances.

2.3 EFFLUENT REDUCTION

Apart from consideration of the impact on receiving environments and the legal aspects involved in reducing effluent, it is clear from a water balance viewpoint that any reduction in effluent implies a corresponding reduction in the required water supply.

It is, however, in the light of increasingly strict legislation regarding effluent discharge, that the reuse of effluent has been encouraged. The Department of Water Affairs and Forestry hold the view that effluent reuse should be considered as a water source(5). It should be noted that only when effluent is released in a way that it is lost permanently to the user should it be considered as a discharge. It is possible to recycle effluent indirectly through discharge and delayed recovery from the groundwater, once infiltration has occurred[6, 7, 8, 9].

3. METHODOLOGY

In order for the above objectives to be met, the relevant water circuits need to be monitored and analysed. The analysis should include an understanding of water flow as well as the relevant contaminant loads. Each water user in the process has certain fitness-for-use criteria which must be met. The water distribution is then altered to assign the minimum quality water, which still meets the fitness-for-use criteria, to each user. Practical limitations and cost implications are then identified and accommodated in a modified practical water circuit. The generation of water and salt balances is covered in detail in a separate literature review but the important points are discussed briefly here.

3.1 WATER BALANCE

The first and essential step in any water reclamation programme is the analysis of the existing water circuits, with the purpose of developing a water balance. Such an analysis needs to specify the required level of flowrate and time scale detail. The accurate and continuous measurement of flow may be costly and all flows which are negligible relative to the total circulation, should be identified and treated as such. A water balance may be based on any level of time scale detail. The choice is dependant on the degree of flow variability and the objectives of the water balance. South African climatic conditions vary seasonally to such an extent that the minimum level of detail should be separate seasonal water balances. In most cases however, a monthly or weekly basis is preferable.

3.2 CONTAMINANT BALANCE

The next step in the water circuit analysis is the identification and monitoring of relevant contaminants, with the objective of developing contaminant balances. Usually however, this is limited to a general total dissolved salt (TDS) balance for two reasons in particular. The determination of dissolved salt concentrations is facilitated by conversion from an electrical conductivity (EC) reading. The measurement of electrical conductivity is simple and can be carried out in situ without the necessity for expensive and intricate instrumentation. Concentrations of individual salts, of particular importance in the mining industry, such as sulphates and chlorides show consistent relationships to the total dissolved salt concentration which implies that many different concentrations can be inferred from a single EC measurement. Each unit operation has different contaminants of concern such as suspended solids in the settling and thickening processes. The important step in developing a useful and practical circuit analysis programme is the up front identification of relevant contaminants or parameters. These may be important for control purposes, regulatory legislation or as indicators of general water quality.

Measured contaminant concentrations are then multiplied together with the flow rates of the water balance to generate the required contaminant balances.

3.3 ASSESSMENT OF FITNESS-FOR-USE CRITERIA

Each unit process utilising water requires a certain quality of water in order for it to operate efficiently and with the minimum of maintenance requirements. These water quality requirements need to be identified and assessed for each operation. Depending on the process, different parameters may be important in terms of the water's fitness for use. The treatment processes related to mining are discussed in individual literature reviews and the relevant parameters for each process can be identified. In general, however, the parameters most often considered are pH, scaling/corrosion indices, total dissolved salt content, suspended solids concentration and, in some cases, microbiolgical indicators such as biological oxygen demand (BOD).

The analysis of each user's fitness-for-use criteria may also serve to highlight instances where existing water treatment is unnecessary or too stringent. It may be possible in such cases to reduce the cost of treatment or to eliminate the treatment altogether.

3.4 IDEAL WATER RETICULATION CIRCUIT

As plant operations are modified, the tendency is to modify the water circuit according to closest availability of water without considering the cost implications of inadequate or superfluous water quality. This practice, together with the historically cheap and plentiful supply of potable water has resulted in operations which have certain unit processes enjoying a better quality of water than is required. Seen from this viewpoint, it is evident that such a situation is not ideal. The ideal water reticulation is one in which each water user receives the poorest quality of water possible, while still satisfying the applicable fitness-for-use criteria, identified in the previous step. In order to illustrate this concept, it is clear that to supply potable water quality to a screening operation for instance, is incurring unnecessary expense. The only applicable criteria for such water would be the absence of scaling or corrosive tendencies and an upper limit for suspended solids content. Potable water, with it's associated cost, far exceeds these quality requirements.

The principle of the ideal water reticulation circuit is thus to assign the worst possible quality of water to each user without compromising that user's water quality fitness-for-use criteria.

3.5 PRACTICAL WATER RETICULATION CIRCUIT

The ideal reticulation circuit has a number of practical limitations. The availability of water is usually limited to a number of sources, each with it's own characteristic water quality. These may include the following:

- rainfall
- collected runoff or seepage
- nearby streams, rivers and dams
- groundwater accessible by boreholes
- aquifers
- potable water
- used process or service water
- collected return water

It is unlikely that each identified water user will be exactly satisfied by one of the above sources, both in terms of quality and quantity of water. Usually, several of these sources are collected together in storage facilities or become mixed together during the process. The result is a reduced number of characteristic water qualities such as raw water, process water and potable water. The next step is then to reassign the various users to one of the available characteristic water qualities, bearing in mind that the fitness-for-use criteria must still be met.

A further practical limitation is the costs associated with extra piping, pumping, head loss etc. These need to be borne in mind when any redistribution is planned. The possibility and cost implications of additional water treatment facilities to improve local water quality must be considered as an alternative. The various water treatment processes are discussed in detail in the individual literature reviews.

Safety aspects such as the possibility of cross contamination between streams, the strict isolation of potable supply and built in water quality safety factors need to be considered.

4. RECLAMATION STRATEGIES

Once a clear understanding of the existing and ideal water reticulation systems has been obtained, the various reclamation options should be explored.

4.1 WATER RESOURCE REALLOCATION

The reallocation of water resources to different users is described in the general methodology above as the final stages in an overall water reclamation and management plan. It is also included here as a stand alone strategy because it is sometimes possible to achieve such a reallocation with little or no impact on the other users in the system.

The benefit to be obtained in such a reallocation is the possibility of reducing water treatment costs, either through a decrease in the degree of treatment required or through a reduction in the volume of water requiring treatment. In some instances, potable water which is purchased may be replaced by an alternative existing source which is available at little or no cost.

In the assessment of user's fitness-for-use criteria and the reallocation of water resources it is imperative that the sensitivity of the user to quality variations is considered and the required safety factors are allowed for.

4.2 RECYCLING AND REUSE

The term recycle suggests that a single user may use the same water several times with or without treatment, whereas reuse refers to the collection of purified effluents and distribution to a different user. The distinction is vague, however, and for this purpose the two strategies are discussed together.

As water is used and recycled there is a concentrating effect on the impurities. In an open system, these may be discharged with the spent water but in a closed system, there is little or no discharge and impurities accumulate and increase in concentration with every cycle of the water. Full recycle implies a completely closed circuit and inevitably some degree of treatment is required in order to remove the collected impurities within the circuit. The cost of treatment needs to be assessed with regard to the cost of increased consumption in an open system. In many cases the latter option is unavailable due to the scarcity of make-up water. The deterioration in water quality usually has negative implications for the user process and distribution system. These problems include salinisation, scaling, corrosion, clogging and erosion. It is possible, however, to reduce the impact of these problems with suitable treatment processes such as desalination, neutralisation, filtration and settling. These particular water treatment processes are discussed in detail in respective literature reviews.

The reuse of water is becoming increasingly necessary as demand on the fresh water supply increases(10),(11),(12),(13). The reuse may be via direct supply after treatment or indirectly via recharge to groundwater. The predominant application of water reuse is in the field of purified sewage effluent. The reuse of sewage effluent for potable purposes is a relatively new field and there is still significant public resistance to the concept. However, there has been a great deal of research in the field and this has concentrated on the epidemiological and microbiological aspects of purified effluent(14, 15, 16, 17). In particular, it is predicted that Cape Town will require a potable treatment facility by the year 2007(1) and a pilot plant has already been erected for this purpose. Actual treatment research has focused on the mass transfer on activated carbon(18) and the use of the oxidants ozone, chlorine and oxygen in pretreatment of the liquid phase. The various disinfection processes are discussed in a separate literature review.

Industries, predominantly the power generation and pulp and paper sectors, are significant users of treated sewage effluent for cooling and process requirements. Other industrial uses are described in the literature(19),(20).

In the mining environment, the treated effluent is being used increasingly for irrigation of slimes dam vegetation as well as for make-up process water and service water(22) at gold mines such as Kloof and several in the Free State Goldfields. The service water application is particularly problematic because of the tendency for underground workers to drink the service water. If the effluent has not been treated to potable standards the danger of infection exists. In some instances, the service water is dosed with chlorine or calcium hypochlorite as a safeguard against such occurrences.

The use of purified effluent for agricultural end uses is more prevalent and has been the subject of detailed research(23, 24, 25). In South Africa, it is estimated that approximately 24 % of the available sewage effluent is used for irrigation purposes, of which nearly 80 % is for crop and pasture irrigation(5).

4.3 WATER RECOVERY

One of the biggest losses of water in any mining circuit is the water associated with tailings disposal. Although a substantial proportion of the water is recovered by the collection and return of pen stock drainage and lateral seepage, the major portion is lost to evaporation from slimes dam surfaces, interstitial retention and seepage to groundwater although this may also be recoverable in the longer term. Many of the older slimes dams were not constructed with artificial underdrainage systems, which further reduces the recoverable volume.

The water : solid mass ratio of a slime with a specific gravity (SG) of 1,45 is approximately 1:1. When this SG is reduced to 1,18 the ratio increases to approximately 3:1. It is clear that a low relative density or SG results in a substantial increase of water volume reporting to the slimes dams. This implies corresponding increases in the losses ascribed to seepage and evaporation. It is likely that the loss due to evaporation is even greater since the slime takes longer to dry and moisture is exposed for longer periods.

The lower slime SGs are more common in the uranium extraction and pyrite flotation processes but poor plant control and spillages can reduce the relative density of any process slime. It has been shown that the extra cost incurred by such occurrences is not only due to water loss but pumping and dam construction costs also increase[26] due to the increase in slurry volume and rate of rise of the dam.

There are many filtration processes which are used in the mining industry for the recovery of water. These are discussed in detail in the relevant literature review.

4.4 EVAPORATION AND SEEPAGE REDUCTION

The phenomena of evaporation and seepage are both associated with the storage of large bodies of water and are thus included in one discussion.

One aspect of evaporation reduction has already been covered in the potential for water recovery from slimes dams. Due to the semi-arid and arid conditions in South Africa, it is necessary to provide ample water storage facilities. Mining areas east and west of Johannesburg as well as the Free State Goldfields are situated on relatively flat terrain. This implies that storage dams are usually fairly shallow, with large surface areas exposed to evaporation. Storage facilities need to be planned in terms of optimum siting and construction should allow for prevailing wind breaks such as tree lines.

It should be remembered that deep storage facilities have the disadvantage of an increased driving head for seepage to occur. A knowledge of the area geology is essential for the correct planning of any storage facility. The problem of seepage can be overcome through the use of dam linings. The degree of lining may vary from well compacted soil and clay layers to impermeable high density polyethylene (HDPE) linings. These are available commercially in a variety of thicknesses depending on the application. Initial capital cost of such a lining is substantial but the often neglected water saving must be considered in any analysis. A further consideration in the choice of lining is the possibility that the constructed dam needs to be desilted on a regular basis.

Although cut-off trenches cannot be included as a seepage reduction measure, it is a form of seepage management which allows the collection and return of the seepage flow. These trenches are found around most slimes dams but the seepage downstream of constructed dam walls is often overlooked.

Historically, slimes dams were not supplied with artificial underdrainage systems and the natural seepage to groundwater was encouraged to enhance the water removal and stabilising process. Modern slimes dams are constructed with artificial underdrainage systems in an attempt to reduce pollution of the groundwater and to enhance the recoverable water volume, albeit at a poor quality.

5. TREATMENT STRATEGIES

It is clear that any reclamation strategy, be it water resource reallocation, reuse, recycling or seepage collection, involves either a poor water quality or a deterioration in existing water quality. The implication is that the water needs to undergo some form of treatment before it can be considered fit for use.

The treatment processes are aimed mainly at reducing high salinity levels by desalination, removing suspended solid content by filtration and settling, water conditioning to reduce scale and corrosion tendencies by neutralisation as well as removal of cyanide and radioactivity concentrations. These processes are discussed in separate literature reviews. In addition, the specific treatment of sewage waste water, boilers and cooling water and the treatment to potable standards are discussed in individual reviews.

If the objective of any reclamation programme is merely to reduce costs, then the cost of extra treatment needs to be assessed in terms of the potential cost of water saving. It should also be remembered that an analysis of the water circuits and user's fitness-for-use criteria may expose areas of unnecessary and over treatment.

6. CONCLUSION

Process water reticulation circuits are usually complex with a high degree of interdependency. Mining environments have the added complexities of underground and domestic circuits. It is thus imperative that any water management and reclamation programme have a detailed water and contaminant balance as it's basis. Analysis of each user's fitness-for-use requirements will highlight possible treatment problems and allow the re-allocation of water resources to more suited users.

The various reclamation strategies need to be assessed in terms of the potential for water saving as well as the possible increase in water treatment costs. Although it is expected that cost consideration will always be the driving force behind water management programmes, the increasing environmental awareness is likely to play an increasingly important role. Indeed, a stage will be reached when the objectives of effluent reduction and water conservation are also economic considerations due to the inevitable escalation in water costs and effluent discharge penalties.

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MONITORING & INSTRUMENTATION

1. INTRODUCTION

The monitoring of water flow and quality is an extensive field and the range of available instrumentation and measuring techniques is immense. It is critical then, in developing a monitoring programme, to identify and clarify the objectives of such a programme at the outset.

The choice of instrumentation for a mining application is often simplified due to the nature of the industry. The hostility of the environment, particularly underground where hot, humid and dusty conditions prevail, makes instrument robustness one of the critical factors. High suspended solid content, large variations in pH and high salinity values are the main factors which make mining streams difficult to handle. These aspects are also, naturally, the focus of water quality monitoring. The measurement of stream flowrates is the first step in any water management program and is essential in determining pollution and solids loading.

The other factor which must be taken into account when defining the monitoring programme is the degree of process control which is desired. Certain instruments are not suited to integration into control loops and this must be recognised. The levels of control vary from occasional reading and recording of visual process indicators to a fully computerised and integrated control circuit characterised by short dead times. In the mining industry many of the required installations are remote and isolated from power supplies and services. The possibility of independent power supply and issues such as signal transmission have to be considered. The theory of automation and process control of unit operations is well documented (1),(2),(3),(4),(5),(6),(7) but is briefly considered in each of the monitoring aspects discussed in this review.

2. FLOW MEASUREMENT

The ideal properties of any flow meter are the ability to accurately measure a wide range of flowrates and the versatility of measuring any type of fluid or mixture. Other desirable properties include ease of installation and instrument robustness, particularly for the mining environment. The range of flow meter types available is extensive and the principles employed in their operation differ markedly. Each flow meter type is particularly suited to a specific application and a basic understanding of the principles involved is essential in understanding the range of possible applications. Because of the large range of meter types there are many flow meter reviews which aim at simplifying the choice(8),(9),(10),(11),(12),(13). In the case of a permanent installation, the decision is made easier because the possible range of flow and stream conditions may be known or at least estimated fairly accurately. When a flow meter is required for temporary installation or test work, the conditions at different test sites may differ widely and the versatility of the instrument is the predominant factor.

Flow meters for pipelines may be broadly classified into two design types: full-bore and insertion. In general, full-bore meters are more expensive but offer a greater measurement accuracy.

2.1 INSERTION TYPE FLOW METERS

As the name implies, these flow meters use a sensing element which is inserted through the pipe wall and into the fluid flow. The element is small in comparison to the pipe diameter and relatively small head losses are caused. An additional advantage with insertion type meters is the possibility of installing the instrument without the need for process downtime, as the element may be hot tapped into the flow line. The accuracy of any insertion type meter is dependant on the ultimate position of the element with respect to the velocity profile of the fluid. Ideally, the element should record the velocity at a point which represents average fluid velocity as well. Generally, insertion meters are less accurate than full-bore instruments but are usually cheaper and have a high degree of measurement repeatability. Many applications, such as dosing control arrangements rely on a comparative measurement using predetermined set points as the control limits. In these cases, it is not essential to obtain a highly accurate flow value.

Many of the insertion type flow meters use principles similar to their full-bore counterparts and these are discussed under the relevant section.

2.2 FULL-BORE FLOW METERS

Full-bore meters usually require substantial alteration to existing pipework, and where this involves expensive downtime of critical operations, this factor needs to be considered. This type of meter is usually threaded into or flanged to an existing pipeline and has an inside diameter equal to that of the pipe. Many of the full-bore flow meters utilise a flow restricting element and the measurement is by means of the differential pressure principle.

2.2.1 Differential Pressure (DP) Principle

Flow meters using this principle employ a restriction to flow, referred to as the primary element, which converts available potential energy to kinetic energy, thereby causing a differential pressure (DP) which is measured on either side of the restriction. This measurement is a function of the square of the upstream velocity as well as the fluid density. Since the density is not usually monitored, a normal fluid density is used but this introduces an error if flow conditions deviate from the assumption. The DP created by the primary element is measured by an electronic DP transmitter which is called the secondary element. These transmitters are considered to be usefully accurate over a DP range of 10:1 which, due to the square relationship between flowrate and differential pressure, translates to a useful accuracy of the flow meter of a 3:1

range between minimum and maximum flowrate. There have been advances in DP transmitter technology which allow a flow range limit of 10:1 but the installed cost may be prohibitive. The various types of full-bore differential pressure flow meters are discussed briefly below.

2.2.1.1 *Concentric orifice plate*

This consists of a sharp square-edged hole in a thin flat plate. The beta ratio, which is the ratio of the hole diameter to the inside pipe diameter, should be between 0,2 and 0,75. Smaller beta ratios produce a higher DP than large beta ratios for the same flowrate. Installation is between special orifice flanges which also house the pressure taps. The orifice plate is supported by extensive experimental work and is fairly versatile. Due to the simplicity of the orifice plate design it is possible to size various plates at different installations to produce a standard DP. This makes it possible to use the same secondary transmitter at all locations. The precision of the fabrication and installation of the plate are, however, critical to the accuracy of measurement and the plates need to be maintained in near perfect condition, which then increases maintenance costs. Highly corrosive or abrasive fluids will also cause rapid deterioration in the plate and result in measurement inaccuracy. Other disadvantages include the high head loss and precise installation requirements.

2.2.1.2 *V-element*

In this design the flow restriction is created by a V-shaped intrusion into the flow path. The distance between the bottom of the V and the pipe wall is called H and the DP obtained for a particular flowrate may be manipulated by changing the H/D ratio, where D is the inside diameter of the pipe. The manipulation of this H/D ratio, analogous to the orifice plate beta ratio, enables the DPs obtained at various installations to be standardised and allows the use of a single secondary transmitter at different measuring locations. Unlike most flow restriction devices, the V element maintains the square root relationship between flow and differential pressure over a wide range of Reynolds number and to values as low as 500. It is thus suitable for flow measurement of slurries, viscous fluids and low flowrates. The design of the V element does not rely on sharp edges and the associated strict tolerances for measurement accuracy. In addition, damage to the upstream face by solid particle impingement is minimised due to the slanted aspect and these two factors make this configuration suitable for slurry flow measurement as well as ensuring a maintenance free life of the meter. Because of the insensitivity to variation in Reynolds number, the V element is able to measure laminar, transitional and turbulent flow regimes without adversely affecting accuracy. This also implies that the restrictions in terms of installation and pipe runs are more lenient than those applicable to the orifice plate.

2.2.1.3 *Venturi tube*

The classical venturi tube has a tapered design in which the inlet pressure is measured at the same diameter as the pipe and a static pressure is measured in the reduced throat section. This diameter is then gradually increased to that of the pipe in the outlet section. A feature of the venturi is the arrangement of several pressure taps in the throat which project radially into the pipe. An average pressure around the circumference is thus obtained. This eliminates the need for long pipe runs upstream and downstream of the meter for flow conditioning. The gradual nature of the flow restriction process and the absence of direct flow obstructions make the venturi relatively maintenance free and ideal for slurry flow measurement as well as applications that require a minimum loss of head.

2.2.1.4 *Target meter*

In this type of meter the primary element consists of a disk fastened to a bar which is in turn connected to the secondary transmitter. The transmitter measures the drag force which results from the flow around the target disk. Although the meter as a whole is fairly robust and the target disk is capable of withstanding solid particle impingement, the bar mechanism is susceptible to clogging and maintenance tends to increase when the solid content of the fluid rises. The target meter is inexpensive, but as with the orifice plate, the required flow conditioning in terms of minimum pipe runs may be impractical and may increase the difficulty of installation.

2.2.1.5 *Rotameter*

The rotameter differs from other DP meters in that the flow area is continuously changing to maintain a constant DP as flow increases. The meter consists of a movable float in a transparent vertical tapered tube and the required DP is just sufficient to balance the gravitational force on the float. The float position then indicates the flowrate by means of a graduated scale on the tapered housing. The rotameter is simply constructed and variations in range are accommodated by changing the float and tube. Usually the tapered tube is constructed from glass or plastic and this limits the applications in terms of pressure, temperature and solid particle content. It is possible to have metallic rotameters but then the float position has to be indicated by means of an electric signal which increases cost.

2.2.2 Other full-bore flow meters

2.2.2.1 *Magnetic*

The principle of operation for the magnetic flow meter is that of electromagnetic induction. Electromagnetic coils generate a magnetic field within the flow tube which is a section of pipe. The flowing fluid acts as the moving conductor and a voltage is thus induced which is picked up by flush mounted electrodes on the inside of the pipe and then measured by the transmitter. Since the fluid is acting as a conductor, it must possess a minimum electrical conductivity, generally 0,1 mS/m. In addition, the fluid must be non magnetic and these restrictions eliminate gases, steam and most hydrocarbon chemicals from the range of applicability. The advantage of this type of meter is the non-intrusive nature of the instrument. Certain magnetic flow meters have electrodes which do not even have direct contact with the process fluid. In this case, the electrodes are mounted on the outside of the pipe, but assumptions regarding pipe wall thickness and degree of fouling introduce significant error to the measurement.

- In the case of the insertion magnetic flow meter the fluid velocity is measured at the sensor which takes the form of an electrode mounted on the insertion shaft. It is recommended that the relationship between the measured velocity at the pipe wall and the average velocity be determined for typical conditions.

2.2.2.2 *Vortex-shedding*

A blunt faced object, called a bluff body, is placed across the line of flow and vortexes or zones of low pressure are formed in the wake of the bluff as the fluid attempts to pass the obstruction to flow. This phenomenon is only apparent under turbulent conditions and the frequency of such vortex formation, which is proportional to fluid velocity, is measured by downstream sensors. The bluff body is subject to erosion and scale build up but can be robustly fabricated. The main limit to application then, is the required minimum Reynolds number for turbulent flow, which varies from fluid to fluid. Installation and maintenance costs are relatively low and there is no need for recalibration.

2.2.2.3 *Turbine meter*

The turbine meter consists of a bladed rotor which is suspended axially in the flow direction and the rate of rotor spin is proportional to fluid velocity. The rotation of the blades can be picked up by magnetic sensors or by fibre optics and reflected light. There are serious accuracy problems associated with air entrainment and slight changes in fluid viscosity. Typical operating problems are the deterioration of support bearings with low lubricity fluids and fluids containing suspended solids. The solid particles also cause abrasive wear on the turbine blades and adversely affect measurement accuracy. This type of instrument has excellent repeatability and can be very accurate when operated under the correct and controlled conditions.

2.2.2.4 *Positive displacement*

The measuring principle in a positive displacement meter is different to other meters in that the volume flowrate is not inferred from a fluid velocity but directly measured by dividing the fluid flow into discrete volumes. The frequency or number of volumes is recorded by a mechanical counter and the meter does not require an external power source for the secondary transmitter. This independence of a power supply is ideally suited to remote and underground installations. The main disadvantage with this instrument type is the requirement for clean fluids with low suspended solids content. The initial costs are high and larger installations require support and adequate foundations.

2.2.2.5 *Ultrasonic meters-Transit time and Doppler meters*

Ultrasonic meters that use the transit time of a pressure wave through a fluid, require a clean fluid as a medium to prevent sound wave scatter by solid particles. This severely limits the range of application and a more versatile instrument is the Doppler meter. An ultrasonic sound wave is projected into the stream and reflected back from entrained particles or bubbles which are assumed to move at the same velocity as the fluid. The Doppler principle predicts a shift in frequency of the wave which is proportional to fluid velocity. This operating principle relies on suspended matter for wave reflection and, in the absence thereof, the stream may be sparged with air bubbles. Transducers are attached to the outside of the pipe and certain pipe materials such as concrete will not transmit the wave. The Doppler meter is effective for slurry flows and only requires a fluid velocity sufficient to keep solid particles in suspension. The successful transmission and reflection of the sound wave is often difficult to achieve with scaled pipes and changing flow characteristics and the Doppler meter is mainly used as a portable instrument for test work which does not require a great degree of accuracy.

To overcome problems associated with non-transmissive pipe materials, the sonic wave principle can be utilised in an insertion type instrument, where the wave is projected axially along the pipe and the average fluid velocity is measured.

2.2.2.6 *Mass flow meters*

In general, mass flow meters utilise the rate of heat loss to a stream, or the rate of temperature increase in a stream, to determine the mass flowrate of the fluid. In both cases, the fluid is passed over a heated body and the rate of energy transfer is then related via conductivity and heat capacity values to determine the mass flowrate. Generally, these type of meters are limited to steam flowrate measurements.

Coriolis massmeters use the Coriolis force which is apparent when an object moves in a system which rotates with an angular velocity. Although these meters have wide applicability, the high installation costs limit their use to difficult fluids and applications which require a high degree of accuracy.

2.3 OTHER FLOW MEASURING TECHNIQUES

2.3.1 Flow Measurement In Open Channels

By employing level measurement over specially designed and constructed open flow channels, it is possible to calculate flowrate as a function of fluid depth. In many cases, fluid flow around a mine is by means of open channel and the flow measuring techniques developed for piped flow are thus unsuitable. The actual means of measuring the level or depth of flow are discussed under the relevant section. The requirements in terms of channel shape and applicable equations are also discussed as these are necessary in the conversion of a level measurement to a calculated flowrate.

There are many different weir shapes and each is designed to cater for the measurement of low, high or variable flowrates. Ease of construction and installation are often required but usually at the expense of accuracy.

Design and calculation details of the range of open channel flow measurements possible are given by (14) but three of the simpler and popular designs are discussed briefly in this review.

2.3.1.1 *Rectangular weir*

This shape and approach can be used at most sites, including rivers, as the river bed can very easily be adapted to a rectangular cross section. If L is the width of the profile and h_0 is the depth of the water then the flowrate is given by:

$$q = 0{,}415(L-0{,}2h_0)h_0^{1.5}(2g)^{0.5}$$

There are various limitations to this modified Francis formula (15) but a high degree of accuracy is possible when flow is correctly conditioned.

2.3.1.2 *Triangular (V-notch) weir*

The advantage of this cross section design is the ability to accommodate a wide range of flow rates. This is then suitable for installation in underground settler feed launders, for example, where the variation in flow rate may be 10:1. If A is the angle formed by the V notch slope and ground then the flowrate is given by:

$$q=(0{,}31h_0^{2.5}(2g)^{0.5})/\tan A$$

where h_0 is again the depth of water. Very often, stormwater channels or penstock return channels are triangular in construction and installation of a level meter is the only modification required for flow measurement capability.

2.3.2 PUMP HOUR METERS

In many cases, pumping of fluid is not continuous but is activated by level control devices at local storage reservoirs or due to demand at downstream receiving facilities. Once the full pipe flowrate has been instantaneously determined by any of the techniques discussed previously, the longer term volumetric flowrates can be established, albeit crudely, by the installation of a hour meter on the pump. This merely records the progressive hours of pumping and can easily be read and recorded on a daily basis by pump operating staff.

3. LEVEL MEASUREMENT

In establishing a complete water balance and for purposes of level control it is necessary to measure and record the change in level of dams, tanks and storage reservoirs. This may be achieved by various methods and the choice of instrument configuration must take into account the nature of the fluid to be measured, accuracy required and site limitations.

In most mining applications, accuracy is not of paramount importance and costly instrumentation such as thermistors and optical devices can be eliminated from the choice. The range of available products as well as a discussion on control applications is given by Turner (16).

3.1 VISUAL DEVICES

Generally, the range of visual devices which include dipsticks, sight glasses and gauge plates are crude methods of level measurement. These methods cannot be integrated into control circuits but are useful as constant and visible indicators in non-critical installations.

3.2 FLOAT ACTUATED DEVICES

The variations in this category can be differentiated by the method of coupling to the level indication system and include chain or tape, lever and magnetic coupling. A variation on the float is the displacer which uses the resultant buoyant force on a partially submerged object to indicate level.

3.3 HEAD DEVICES

The use of hydrostatic head as a level measurement is one of the more common principles involved but the configurations are varied and site specific. The measurement of differential pressure, as with the range of flow meters utilising this principle, is very dependant on fluid characteristics and this is often too variable for sensible level measurement.

3.4 OTHER METHODS

Among the wide range of alternative methods are capacitance and conductive-type meters which utilise the electrical characteristics of a fluid to measure depth. In applications where fluid contact is not recommended, an ultrasonic method, which relies on reflection of a sound wave from the fluid interface, may be used. It is usually this type of level detection which is used in conjunction with a weir construction for the determination of flowrates as discussed in 2.3.1

4. WATER QUALITY MONITORING

The measurement of flowrates is useful in determining quantities of water used, required and discharged in various processes but certain water quality parameters are essential in the optimisation of such processes and in the determination of pollution loads and the effect on process equipment.

4.1 pH MEASUREMENT

pH adjustment of acid and alkaline waters is required for various reasons such as prevention of corrosion by acidic water, prevention of scaling by high pH waters and pH correction prior to a downstream treatment process. In terms of underground settling, for instance, flocculants operate optimally over a, sometimes narrow, pH range and flocculant addition at incorrect pH levels results in inefficient settling and increased flocculant consumption. pH adjustment is achieved by various methods and the choice of method is critical to correct positioning of the pH monitoring installation. The most common neutralisation chemical is lime which reacts slowly. Rapidly reacting reagents such as soda ash and caustic soda are used to a lesser degree. Any pH monitoring system must be positioned such that sufficient reaction time has been allowed for. However, at the same time, the distance between monitoring and reagent dosage point cannot be too great, as this would result in a large dead time, and the loss of effective control.

In the field of process pH monitoring the pH sensitive glass electrode is the dominating technique of measurement. This type of instrument requires a reference electrode which is not sensitive to pH and which measures the output of the glass electrode. The theory of the chemistry for pH measurement is discussed in various articles (17, 18) and the on-line application of pH measurement is discussed by Hulley (19).

The development of the pH and particularly the reference electrodes has mainly been due to the requirement for a more robust form of instrumentation. The types of electrode assemblies are varied but aim mainly at avoiding contamination of the electrodes by the process fluid. The main electrode holder arrangements are the flow-through cell, in-line installations either in a bypass loop or directly hot tapped into the process line and dip systems which may be used in open tanks and channels. This is the most suitable arrangement for installation at the underground settler feed launders. As discussed previously, flow in these launders can vary tremendously and the instrumentation has to cater for the possibility of complete immersion. Correct design of the weir can also eliminate the possibility of undesirable electrode exposure and baffles may be used to ensure sufficient mixing in the measurement region.

One of the main problems involved in on-line pH measurement is the contamination and fouling of the electrodes and a wide range of automatic cleaning systems are available. Among the techniques used are brushing, jet cleaning and ultrasonics but probably the method which has most applicability in the hostile mining environment, is a form of abrasion using plastic balls contained in the mixing chamber. Another alternative which is less practical but increases the electrode life is that of maintaining the electrodes in a flushing solution for long periods between measurement. Obviously the continuity is sacrificed, but in systems where large fluctuations are not common, this operation may be possible.

The sensitivity, rangeability and non-linearity of pH measurement is such that a high degree of accuracy is not possible and McMillan (18) discusses possible sources of error and potential solutions.

4.2 TURBIDITY AND SUSPENDED SOLIDS

The on-line measurement of suspended solids content is not a common practice in the mining industry and jar settling tests are the norm. Flocculant dosage is usually at the discretion of the operator and is not able to take account of the massive variations in hydraulic and solids loading which occur at underground settlers. This results in either the excessive and unnecessary consumption of flocculant or the inefficient operation of settlers and usually a combination of both.

In this situation a feed-forward control is ideal. A suitable system would comprise an on-line turbidity meter upstream of the settler or thickener. In conjunction with information from a flow meter, the controller would then assess the flocculant dosage requirements for a particular solids loading. This, together with the required pH adjustment, would ensure optimum settler operation.

Although the instrumentation is available, the correct control of underground settlers is not common. Similar systems, using streaming current detectors for coagulation control in water treatment plants, are discussed in the literature (20, 21).

The solids content in settler and thickener under flows is usually determined by weighing a known volume of the mud and comparing it to the expected equivalent weight of clear water. This method implies that the mud under flow removal from such settling processes is batch operated. The continuous and on-line measurement of liquid density, and thus specific gravity, is achieved through various methods which are discussed by Hoeppner (22).

4.3 TOTAL DISSOLVED SOLIDS

The salt load on receiving waters from the mining industry is often substantial and with the increasing trend towards water reuse and recycling within a mine this salinity can reach elevated levels. It is essential, as a management tool, to determine the source of dissolved salts, define the salt loads at strategic points in the water reticulation system and attempt to reduce the salt content leaving the mine in effluent discharges.

Although it would be ideal to obtain a complete analysis of the relative contributions of various salts to the total salt load, this is not feasible on a continuous basis. Such an analysis must be restricted to a periodic sampling which is dictated by objectives and financial consideration. An estimate of the total dissolved solids (TDS) content of streams is possible by means of electrical conductivity monitoring and the application of a linear and usually constant factor enables conversion to TDS.

5. CONCLUSIONS

The correct control of processes is becoming increasingly important in the mining industry in an attempt to continually reduce operational costs. The first step in any process control and management programme, once the objectives have been defined, is the effective monitoring of key parameters.

The measurement of volumetric flowrate and reservoir levels, at critical points in the water reticulation, is essential in determining a water balance for the operation. This balance is the tool with which precious water resources may be managed. Unnecessary or avoidable losses may be identified and loss prevention strategies may then be implemented. The measurement of certain water quality parameters such as pH, turbidity and total dissolved solids content is necessary to improve operation of settling and thickening processes, as well as reducing maintenance costs due to corrosion and scaling of process equipment.

In general, mining applications do not require a high degree of accuracy from the instrumentation and the main consideration is usually one of instrument robustness for hostile underground environments. Due to the remote nature of many surface installations the transmission of data and independence of power supply are the main considerations. It is the lack of consideration of these factors which has often resulted in the failure of previous monitoring programmes. The definition of objectives and identification of site limitations are crucial in developing a sustainable and useful monitoring programme.

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GROUNDWATER

1. INTRODUCTION

Groundwater is becoming a very important resource in South Africa, and pressure on the aquifers, through domestic use and pollution, is on the increase.

Impacts on aquifers affect this resource in two specific ways. Firstly through lowering the water table by pumping out excessive amounts of groundwater for domestic use, agriculture or mining, and secondly by contamination of groundwater by domestic and industrial waste, septic tanks and associated facilities, agricultural chemicals etc.

During recent television programmes, an official from the Department of Water Affairs, estimated the regional drop in the groundwater table, in the Northern Transvaal, at 20 - 30 meters. The drop in water tables is a combined effect of drought and over-exploitation of the resource. This drop in the water table is apparent in some of the vegetation in the area, and specifically in the leporine zones and riverine forest of the Olifants and Limpopo rivers.

Although aquifer contamination may not have as drastic a short term effect as the drop in water table, the persistence of the contamination may make remediation almost impossible. An example is the effect of underground fires in the Eastern Transvaal Highveld. Heat from coal continuously burning underground speeds up pyrite oxidation, which increases acidity and salinity of ground and surface water. Currently ecosystems like Loskopdam are being affected, while agriculture further along the Olifants river will probably be affected, followed by the Kruger National Park and eventually areas in Mozambique.

South Africa's most important aquifer comprises the dolomitic rocks of the Transvaal Sequence, and many of the mining operations are located in or close to these strata, with both dewatering and contamination problems. Of specific interest is the dewatering of the dolomite compartments overlying the gold-bearing formations. Results have been the depletion of an aquifer of major regional importance, sinkholes and subsidence(12). Due to the complexity of the mining impacts on groundwater, this review should be read in conjunction with the reviews on water sampling, discard management (residue deposits) and undergroundwater management.

2. CHARACTER AND EXTENT OF GROUNDWATER IMPACTS

The duration of impact can be divided into long term and short term impacts. Surface waste dumps and subsurface mining operations have an impact on hydrochemistry in the long and short term. Short term hydrochemical impacts associated with coal mines result from direct solution and base exchange reactions between water and host sediments. The severity of short term pollution hinges mainly on quantities of contaminants present in overlying rock and in mine workings.

Base exchange reactions involve substitution of monovalent cations, particularly Na and K, by bivalent cations particularly Ca and Mg. The result is an increase in the Na and K content of groundwater with the corresponding proportional decrease in Ca and Mg. Ions that dissolve directly into groundwater and do not require a process of base exchange, are another cause of short term pollution (e.g. NaCl, $CaCO_3$). The result of direct solution is an increase in TDS in the groundwater.

In the long term, acid mine drainage has a hydrochemical impact on groundwater, where mining has occurred. Acid mine drainage is a well documented process. Pyrite oxidation forms ferrous iron and sulphuric acid, ferrous iron is further oxidised to anhydrous iron oxide and more acidity. Pyrite oxidation is related to: pyrite, oxygen and ferric iron concentrations. Certain bacteria catalyse pyrite weathering by catalysing ferrous iron oxidation to ferric iron and thus speeding up the process of acid formation. The acidic water produced by acid mine drainage, will increase dolomite groundwater compartment weathering. The acidic water dissolves the $(Ca/Mg)CaCO_3$ that dolomite consists of, resulting in compartment collapse and sinkholes.

A secondary groundwater aspect of coal mining, is combustion of coal underground. The fires spread and create cavities in which water is brought into contact with sulphates. The magnitude of the problem has been pointed out earlier.

Open cast coal mining is often situated in valleys where the depth of overburden to coal is small. Mining in or through valleys necessitates dealing with the increased fissure water (due to a water table closer to the surface in valleys), and two problems present themselves. Increased groundwater influx into the pit and groundwater entering the open cast working of the backfilled voids of the mine become saline and need to be contained and dealt with in an environmentally acceptable manner.(11) Some precautions are discussed at the end of this report. Open cast mining overburden is replaced in mined-out areas. The replaced overburden results in higher hydraulic conductivity and porosity.(12)

In terms of groundwater, gold mining impacts show some similar characteristics to coal mining. Waste disposal seems to be the primary factor of concern, followed by dewatering. Of particular importance in gold tailings, is cyanide. Attenuation and degradation mechanisms of cyanide can be classed into those effective in aerobic environments (unsaturated groundwater regimes) and those effective in anaerobic environments (saturated groundwater regimes).

Aerobic environments:

Volatilisation - of hydrogen cyanide to free cyanide occurs primarily when water with a pH less than 7 is exposed to open groundwater systems. Temperature, pH, solution concentrations and aeration affect the rate and extent of volatilisation.

Oxidation - of HCN and CN^- to cyanate or hydrogen cyanide typically requires a mineralogical, bacteriological or photochemical (sunlight) catalyst.

Biodegradation - micro-organisms metabolise the cyanide and biologically catalyse the direct oxidation of CN^-, thereby consuming free cyanide to generate cyanate

Photo degradation - occurs in surface water exposed to sunlight and mainly involves the dissociation of iron cyanide when exposed to light.

Anaerobic environments:

Adsorption - of free and complex cyanide by clays, and organic matter. Soils with a high anion exchange capacity, such as kaolin, can attenuate negatively charged cyanide ions and complexes. Metal cyanide complexes can be readily absorbed.

Chemical complexation - cyanide can be removed by the reaction of an insoluble simple cyanide with excess cyanide ions in solution to form metal cyanide complexes.

The last two mechanisms may be temporary, changes in pH, composition and concentration of surrounding waters will affect the stability of the attenuating mechanism.[8]

3. CASE STUDIES

Because of the many variables that may influence groundwater, problems vary from site to site. A few case studies will be discussed to show some groundwater characteristics.

Inflow of groundwater into Grootgeluk mining area is continuous below 40 m; this inflow has a high TDS content of 2 000 - 8 000 mg/ℓ. The main driving force behind the TDS was found to be calcium, magnesium, and particularly sulphates[6]. A study of the influence of coal mining on water quality in Canada showed some corresponding characteristics. High TDS, fluoride, sulphide and sodium concentrations, as well as nitrate and iron has been documented.

Following a study of groundwater at Majuba Colliery, the following has been established and published. Two main groundwater systems, with distinct water chemistries were identified: A shallow system, with a calcium-manganese bicarbonate character, and a deep system in dolorite sills with sodium as dominant cation. High alkalinity (pH>9) and fluoride concentrations were also persistent.

As discussed earlier, underground fires increase the groundwater salinity problem associated with mining. A publication on a project aimed at preventing the spread of an underground fire from Middelburg steam colliery to the adjoining Coronation colliery near Witbank, has shed some light on possible precautions that can be applied. Flooded mines and progressing underground combustion, makes the necessary "surgical blasting" of the overburden very difficult in this area. This article illustrates that, by combining two processes - coal production and rehabilitation - into one operation, time and money are saved. This also limits degrading of original soil quality, which tends to degenerate if left unattended.

Gold mines situated in the Free State Goldfields also showed some groundwater salinity problems but the characteristics in the area differ from the coal mines in the Transvaal. According to a WRC report on the OFS Goldfields, groundwater levels are topographically controlled and groundwater in the Ecca sediments shows a poorer quality than those in the Beaufort sediments. This can be ascribed to the mixing of connate water of marine origin with meteoric water, thereby explaining the high salinity of the groundwater from the Ecca Group.

Sodium, chloride, calcium and sulphate concentrations were of concern followed by cyanide (and some metals that were less of an environmental threat in this study). Ionic ratios of Ca/Cl, Na/Cl and Ca/Na were used to hydrochemically characterise the water. Mines in operation for about 40 years on Ecca sediments show pollution restricted to six kilometres from disposal sites, while mines in operation for about 10 years show pollution limited to 1 kilometre from disposal sites, on Beaufort sediments.

It was observed that Cl^- (chloride ions) in the OFS area spread to a greater extent than any other of the relevant ions, and due to its inherent characteristics it is used as a conservative tracer. Concentration of Cl^- can be used as a conservative estimate to determine the maximum spread of pollution. Heavy metals have elevated concentrations in close proximity of slimes dams that tend to generate acid mine drainage. Reverse ionic exchange in the vicinity of slimes dams also causes groundwater to become enriched with calcium. Monitoring of groundwater around disposal and monitoring sites as well as locational analysis for waste disposal are suggested as management options.(16)

In contrast to the OFS studies, reports show that of the 50 000 t/yr of salts discharged into near-surface aquifers from mine dumps on the central Witwatersrand, sulphate is the major contributing factor.(12)In a local study, it was found that shales on the West Rand restrict gold mine slimes dams seepage, to groundwater, effectively to less than 1 kilometre.(17)

4. MONITORING AND MANAGING GROUNDWATER QUALITY

Some reports suggest groundwater quality and water table monitoring as a management option(16). Though monitoring is very important for management decision making, it is only a tool to identify impacts, thus the results of the monitoring should determine the eventual option followed.

A major factor, for determining the impact of mining on groundwater, is described as draw down, and not so much water quality.(4) A practical method for groundwater volume determination, is by continuous pumping from a borehole and determining the drop in water level and time of recovery to initial water level.(4) The influence of draw down has been discussed, but practical alternatives are limited. For this reason quality impacts will still be important during monitoring and modelling for remediation and pollution control.

Monitoring and mapping of aquifer susceptibility to pollution is considered a critical first step in implementing groundwater management programs. GIS's have been used in many aspects of groundwater management and modelling. Spatial models designed to evaluate groundwater vulnerability to contamination have been more widely implemented in GIS than any other single type of groundwater related model.

Erdas and ARC-INFO software has often been used in the USA with a groundwater model called DRASTIC. The acronym is derived from seven factors considered in the model: Depth of water table, Recharge, Aquifer geology, Soil texture, Topography, Impact on vadoes zone and Conductivity (hydraulic) of aquifer.(19) Because this model takes all these factors into account in combination with the GIS capability, it is a powerful groundwater management tool.

Another application in this field is the monitoring of waste sites that could affect groundwater. Geological remote sensing can be used to characterise solid waste landfills and hazardous waste sites in two ways: the use of digitised stereo photos to compute digital elevation models (DEM) of the landfill and the use of multispectral image data to map chemical compositional differences that are environmentally significant. The former can be used in water run-off prediction. The latter can likely be used to map clays , ferric oxides (Landsat band 5 & 4 (0,5 - 0,7 µm)) associated with contaminated groundwater geochemical "cells", stressed vegetation, and gaseous plumes escaping from landfill.

Ferric oxides and hydroxides (such as hematite, limonite and goethite) can be mapped with multispectral scanners and are often found were water has discharged through gravel and sand. Under some circumstances the oxidation-reduction phenomena associated with contaminant leakage into groundwater will deposit iron oxides in a manner similar to ferric and ferrous oxide exposures related to geochemical cells related to mineral exploration. This is an indirect approach similar to that used by exploration geologists, who map clay and sulphate exposures when looking for gold deposits that contain gold in quantities too small to detect directly.

It is also likely that surface exposures of exotic compounds with unique spectral features, such as cyanide and chromium, will result from leachate or contaminant interactions with soils and water. An example is "blue soil" associated with cyanide compounds.[20]

If groundwater contamination is characterised by means of monitoring, the management options to be followed can be identified for specific problems that exist. Characterised contaminants are exposed to alterations that can complicate modelling and monitoring. Contaminants passing through groundwater systems are acted on by a range of different processes, which can be conveniently represented as four filters:

- Mechanical filters - sand, gravel and stone may block the flow of some contaminants.
- Physical filters - reduces contamination through sorption in sediments.
- Chemical filters - clay and humus exchange ions with those contained in water.
- Biological filters - microbial activities alter and deplete contaminants.

These filters usually act together and interactions are strong. Aquifers are commonly sub-divided into intergranular (porous) and fissure flow (karstic). In porous aquifers, all four filters are important, and a great deal of research has been done on these aquifers. Less is known about the processes controlling contaminant cycling in karstic systems. Because of open crevices, physical and biological filter effects predominate. In porous aquifers, the unsaturated (vadose) zone plays an important part in retention and self-purification of contaminants. Flow properties of karstic aquifers are different inside and outside drains, a phenomenon referred to as directional hydraulic conductivity. For non-interstitial karsts, contaminants are estimated more easily, when water is renewed frequently.

As for surface water, improving conditions for self purification is a possible management option as pollution prevention is. Self purification, is less readily achieved in some aquifers like karsts. There is usually not enough time for the water in karst formations to be thoroughly oxygenated before it reaches the saturation zone. Self purification in the saturated zone can therefore only take place anaerobically. The low density of micro-organisms reduces the rate of mineralization of organic substances. The almost total lack of algae inhibits the conversion of nitrates when they are abundant. Biotransformation and biodegradation is not well researched in karsts and more study is needed, this is particularly true for South Africa's dolomite aquifers.[13]

As groundwater characterising is dependent on monitoring results, some factors affecting sampling results are discussed.

Contaminants in groundwater can be attenuated through absorption or enhanced through scavenging. The original composition of the pollutant may be greatly altered between the source and the sampling point. The source and extent of the pollution is therefore not easily determined. Pollution source characteristics or "signatures" which are not modified between source and receptor (sampling point) are therefore desirable for monitoring. Trace elements are good candidates for the signature approach, because even if their chemical forms and absolute concentrations change, individual elements with similar characteristics are expected to be affected equally. Solute tracers, micro-spheres and labelled bacteria (DNA specific fluorescent marker) have been used in studies to examine hydraulics of aquifers.

Instrumental neutron activation analysis (INAA) and element ratio's are used to indicate pollution sources. Relative contribution apportioning was also accomplished. Inductively coupled plasma-mass spectrometry is also said to be suitable for these purposes. Cost effectiveness or economical factors are not discussed. Apportionment of groundwater contamination, between bordering slimes dams or mines, could probably be resolved at a lower cost in this manner than in a legal dispute.

Groundwater element ratios have been used to indicate relative contributions to groundwater pollution from slimes dams in the Far West Rand.(14)

Radioactive isotopes used in groundwater studies, often offer the ability to trace groundwater pollution back to sources of pyrite oxidation. Radiogenic strontium and lead in water have isotope signatures which are determined by the geology of the source material. Lead is associated with sulphide minerals and exhibits varying isotopic signature depending on the age and type of sulphide mineralization. Lead isotope ratios may be utilised to estimate relative proportions of pollutant mixing in water systems contaminated by two different sources(21). This can be applied to apportioning contributions of acid mine drainage from different discard dumps in a area. Examples in gold and coal field areas exist.

5. POTENTIAL HEALTH AND ENVIRONMENTAL THREATS OF CONTAMINATED GROUNDWATER

The threats of groundwater do not differ much from those of surface water. There are two elements that do however seem to be of more concern in groundwater than in surface water.

Because of the absence of sunlight and reduced oxygen, activity of bacteria that break down nitrates, is restricted. Methemoglobinemia - an acute toxic response to nitrate exposure that precludes transport of oxygen by the blood, has been linked to death of infants after intake of groundwater with high nitrate concentrations. The relationship between nitrate intake and cancer has not been conclusively demonstrated, but some evidence points to a linkage. Infants 6 months and younger are at highest risk because of high fluid intake : body weight ratios and a higher stomach pH.

The other constituent that proves a potential threat in groundwater is cyanide, as photo degradation underground is absent. No publications on lethal concentrations were found, but health can be affected, and warnings are usually placed on boreholes and wells in the vicinities of slimes dams.

6. REMEDIATION OF CONTAMINATED AQUIFERS AND TREATMENT OF GROUNDWATER

Remediation is usually expensive, so pollution control and precautions should be applied where possible to prevent situations where remediation may be necessary.

Pump and treat methods are discussed in some literature, where water is pumped to surface, treated and pumped into the aquifer again. In situ biological treatment and granular activated carbon (GAC) seem more practical, but expenses are still not justified easily. The GAC method leads to good lead, nickel and cadmium removal, as stipulated in CERCLA (comprehensive environmental response compensation and liability Act of the USA). Hazardous substance ion exchange was tested, reverse osmosis for removing radon and aeration is being tested too. Radon removal by GAC and uranium removal by reverse osmosis and ion exchange is also investigated. Costs of GAC are said to be $ 0,5 - 0,8/ gallon, but this could vary according to contamination.[1] Other processes like powdered activated carbon, (PAC) are also referred to in articles about groundwater remediation.

Unique groundwater techniques for nitrate reduction seem to have been developed which are less costly than conventional methods. Groundwater denitrification using a USB (up flow sludge blanket) reactor, with ethanol as the carbon source, indicates effective treatment of contaminated water. This type of reactor was found to be strongly affected by the hardness of water.

In a South African article about groundwater treatment alternatives, the hardness element of this country is discussed. Calcium and magnesium hardness exists as positively charged ions (Ca^{++} and Mg^{++}). Technically hardness also includes iron (Fe^{++}), manganese (Mn^{++}), and other polyvalent cations. However, these ions are usually present only in small amounts and can be ignored. The soap-destroying and scaling properties of hard water often justifies several treatment methods. Dolomite geology is a major source of calcium and magnesium ions in South African groundwater and dolomite is one of South Africas most important aquifers.

Calcium and magnesium are most soluble in acidic water, and are usually in a bicarbonate form. Hardness may be classified as temporary (carbonate) or permanent (non-carbonate). Carbonate hardness can be precipitated by boiling. Non-carbonate hardness, associated with sulphate, chloride and other anions, cannot be reduced by boiling.

With $CaCO_3$ concentrations < 60 mg/ℓ, water can be classified as soft while concentrations > 120 mg/ℓ are classified as hard. Hardness concentrations of > 1 000 mg/ℓ, in dolomitic groundwater associated with gold mining, are not uncommon in South Africa. The economic consequences of scaling will be felt over time if the condition is not addressed. Methods for softening groundwater include; cold and hot-process precipitation, ion exchange, reverse osmosis and distillation.

Cold-process precipitation softening - soda ash or lime is added to let calcium and magnesium settle out, leaving the water softer.

Hot-process precipitation softening - the same process as in cold softening is used but it is faster and more effective. Silica may be removed along with calcium and magnesium.

Ion exchange, reverse osmosis and distillation are discussed in more detail in the review on desalination.

7. PREVENTION AGAINST CONTAMINATION

Open cast coal mining is often situated in valleys where the depth of overburden to coal is small. The water table usually resembles the topography to a certain extent, but in valleys the water table is situated closer to the surface. This shallow groundwater table is therefore often intercepted during open cast mining in valleys.

Open cast mines can minimise groundwater entering workings by dewatering boreholes around the pit and by trenches that act as river diversions, but will also intercept groundwater. The costs of trench excavations may under specific circumstances be less than pumping from the workface. It seems that this method might be of specific use in the open cast mining sector.[10]

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RIVER DIVERSIONS

1. INTRODUCTION

Disturbances of water courses as defined by The Aide Memoire for EMPRs by the Department of Mineral and Energy Affairs (RSA), includes disturbances of natural flow or alignment of the water course in any way. A permit in terms of section 20 of the Water Act, 1956, to alter the course of a public stream may be required before an EMPR is approved.

Depending on the importance and timing of the diversion and its potential impact on the environment, information may be requested. The information requested relates to topographical plans indicating alignments, linings, erosion control, hydraulic structures, storm water entry and floodlines(1). This information must relate to the final situation upon closure since approval for temporary diversions, not showing the final situation, will not be given.

Water tightness and structural stability of diversions need to be estimated, and catchment yield alterations must be considered continuously.

2. PURPOSE OF RIVER DIVERSIONS

2.1 DIVERSIONS TO PREVENT SUBSIDENCE IN DOLOMITIC AREAS

As many of the gold mining activities on the Witwatersrand and West Wits line are associated with dolomitic areas where sinkholes and subsidence occur, extensive piping, canalising and diverting of natural water courses has taken place in some areas (e.g. north of Westonaria). This could have an impact on the ecology of the area, but in this way the impact can be limited and controlled, while subsidence and sinkholes could destroy buildings and roads with loss of life, and also adversely affect riverine ecosystems.

The restoration of the water table in mined areas is another method of decreasing the possibility of surface movement and subsidence. Examples of such restoration of the water table, in the Western Areas region, show that by pumping water back into sinkholes and boreholes, ground movement can be limited(5). Back pumping into sited boreholes is advised as flow within and through compartments can be reduced in this manner.

2.2 DIVERSIONS TO PREVENT SHAFT FLOODING

As mining of the Witwatersrand gold reef outcrops ceased in the East Rand Basin, little or no sealing of workings occurred, and no provision was made to prevent surface waters from flowing into them (6).

Serious flooding of underground workings were experienced in January 1972 at SA Lands and Exploration and Vlakfontein. The flooding was ascribed to 4 different reasons.

1) Alterations to the watershed - Jan Smuts Airport construction led to the drainage system being altered from a northerly run-off to a course south-east, towards Benoni. Through construction of motor ways and freeways east of Johannesburg considerably larger quantities of rain water flowed into the Blesbokspruit.

2) Erosion of slimes dams, silted up natural streams, reducing the flow rate, causing wider channel of flow.

3) The widened flow was slowed down and spread further by dense growth of reed beds.

4) Torrential rains, much higher rainfall figures than usual were recorded (6).

Sinkholes also occurred during this time. After the flooding of the mine workings, by an estimated 250 - 2 500 Mℓ of water, an investigation of the entire river canal was undertaken. From the Kleinfontein dam the overflow initially flows in a 500 long meter concrete-lined canal, it was found that cracks in the cement and trees growing in the canal caused water to leak and overflow into abandoned mines. The canal enters into a marshy area were water entered old abandoned mine workings, from there the water enters a 400 meter long concrete lined canal. Further downstream, other wetland areas were silted up and flow would have been widened more.

The end result was having to remove many obstructions from the waterway and an adequate waterway had to be re-established by restoring the old water course to its original path(6). Blasting was used in some areas to establish drainage channels.

2.2. DIVERSIONS FOR FLOOD PREVENTION IN OPEN CAST MINING

As far as coal mining in South Africa is concerned, open cast mining tends to concentrate in valleys, as the overburden here is less. This however causes an increased influx of ground and surface water and flood risk. Often mines use cut-off trenches and river diversions to reduce influx and flood risks.

Canal construction is also often used to divert water around open cast mining or shafts. The scale of diversion can be up to 40 meter deep trenches of 880 meters long(17). These canals alter the regional hydrology considerably and should be planned and maintained with this in mind.

Wagner (1994) discusses a river diversion at Optimum coal mine, which reduces chemical contamination of water drastically, but the effect of the diversion into the adjacent drainage basin is not addressed(20).

This factor needs to be kept in mind for impact assessment purposes, as not only the river diverted is affected, but the inter basin transfer impacts on the receiving drainage system as well. South Africa has one of the lowest conversions of rainfall to runoff relationships (8,6 %) in the world(14). Water resources are unevenly distributed with one third of the country yielding a mere 1% of the total runoff. In addition rainfall is invariably either equal to or less than potential evaporation. In many mining areas, however, the majority of precipitation takes the form of thundershowers, which lead to fluctuating peak flow run-offs that need to be monitored and managed well.

Detailed research into the ecological impacts of inter basin transfer schemes, is virtually non-existent both globally and nationally(14). Growing awareness of the serious nature of such impacts - "loss of biogeographical isolation, loss of endemic biotas, introduction of alien or invasive species, change in water quality and alteration of hydrological regimes" - demands an urgent appraisal of current planning and research strategies(14). In a paper by Petitjean & Davies (1988), four Southern African transfer schemes (Orange river Project, Tugela-Vaal Scheme, Eastern National Water Carrier and Leshoto Highlands Water Project)are considered and discussed.

Major river diversions as in the Great Fish River, have lead to considerable alterations in the aquatic ecology. Data on the hydrology, chemistry and invertebrates of the Great Fish River prior to the opening of the water transfer scheme, was compared with similar data for post transfer conditions. The mean annual runoff in the upper river was increased by around 500 %, and highly mineralised water of the Fish river was diluted, reducing concentrations of sodium, magnesium, chloride and sulphate but not calcium or total alkalinity. The invertebrate communities of the riffles were changed substantially(15).

3. IMPOUNDMENTS AS FLOW CHANNEL ALTERATIONS

Studies of smaller dams in catchments such as the Olifants river (Eastern-Transvaal) show that small dams (farm dams) can have an impact on the river system(2). For this reason, most dams - even if they seem insignificant - need to be assessed by means of a permitting system.

During droughts specifically, the impact of farm dams on the inflow to major reservoirs give rise to much concern. In the Olifants river catchment, it was found that yields from four major reservoirs in the study area had been reduced significantly by the minor dams. In order to assess the influence of small impoundments a computer model has been used, to simulate three conditions by changing the land use: natural conditions; 1987/88 hydrological year; and 1987/88 hydrological year with all minor dams removed. In this way the impact of small dams on the catchment could be determined. Computer models, satellite image processing and specifically Geographical Information Systems (GIS) are becoming very important in hydrological management and modelling. A separate literature review on GIS and database management has been prepared for this project, in which the applications and possibilities of GIS in water management are outlined.

On the Witwatersrand and specifically the south of Johannesburg, the urban, industrial and residential sectors of the city have encroached upon established mining activities. The dams and hydrological structures on these mines often have to deal with highly altered stream flow patterns as infiltration in urban areas are usually drastically reduced, causing large volume run-offs in short periods of time. Stewart Scott Inc., together with Pulles, Howard and de Lange Inc. and the Johannesburg City Council, are currently investigating the identification, causes and management of typical pollution problems encountered in lakes and dams in urban areas(3). Urban impoundments that were initially used for mining purposes, and are now safely and extensively used for recreation (e.g. Florida Lake and Wemmer Pan) are also being studied.

4. MANIPULATION OF FLOW REGIMES

The adjustment or naturalisation, of stream flow is an essential step in any water resource analysis, since it provides a baseline against which developed catchments can be compared. If records of discharges and abstractions are available, the naturalisation of a stream flow record is not difficult to produce. Reliable records of abstraction and discharge are however, not usually readily available. This makes the operation a very difficult one and often estimations are involved, which results in less effective decision making.

The abstraction of water from rivers for mining, industrial and domestic purposes and the discharge of effluent have also had a significant impact on the hydrological regime of the Olifants river(2).

There is growing international concern, regarding the increasing rate of deterioration of natural environments of rivers. In the past, much of this concern has focused on water quality problems, and only recently, has the manipulation of flow regimes been recognised as a major factor affecting the health of rivers(4).

A synthesis report of methodologies presently available for instream flow assessments has recently been released by the Water Research Commission (WRC). This work entitled Assessment of Instream Flow Incremental Methodology, and Initial Development of Alternative Instream Flow Methodologies for South Africa, emanates from the project "Assessment of instream flow requirements of rivers".

The Instream Flow Incremental Methodology (IFIM) was applied and tested in a South African situation which presented the local realities and problems of assessing the water requirements of rivers. The research showed that neither IFIM, nor the analyses of daily flow records obtained from the Department of Water Affairs and Forestry (DWA&F), provided the means to produce a recommendation of a comprehensive modified flow regime for a regulated river according to the requirements of the DWA&F.

IFIM was devised by the United States Fish & Wildlife Service. It comprises a collection of analytical procedures and computer programmes and includes the *Physical Habitat Simulation Model*, PHASBSIM II, which is its best known component. In its entirety IFIM is said to evaluate the effects of incremental changes in stream flow on channel structure, water quality, temperature, and availability of this physical micro habitat for selected target species. Availability of this physical micro habitat, which is simulated by PHABSIM II, is actually the principle product of applying the methodology, and is used in negotiations for an acceptable discharge or modified flow regime for the river of concern. Channel morphology, hydraulic characteristics of flow through channel sites, water depth and velocity, as well as substrate, hydraulic and vegetal cover over a range of discharges is taken into account during the procedure performed. The substrate and cover conditions together are termed the channel index (CI).

The output from the model consist of plots of Weighted Usable Area versus Discharge (WUA-Q) for each target species. These predictions illustrate both present weighted usable area and, future availability of weighted usable area after such a development.

However according to the report the development of an alternative methodology has started to evolve, for situations where there is little time, money or data available for such research. The two new methodologies are being developed in liaison with other river scientists and DWA&F personnel.

Method One uses cluster and discriminant analysis. Method Two used correspondence analysis and covariance biplots. Method One groups rivers that have similar hydrologic characteristics, while Method Two groups rivers that are within defined geographic regions (the winter rainfall region of the SW Cape, seasonal South Cape and the summer rainfall region). Rivers were further subdivided into smaller geographical regions by method two.

The Building Block Methodology is based on the assumption that species occurring in a river can cope with base flow conditions that naturally occur in it often, and may be reliant on higher flow conditions that occur in it at certain times, such as specifically timed floods. It is also understood that certain kinds of flow influence channel geomorphology more than others, and that incorporating such flows into the modified flow regime will aid maintenance of the natural channel structure(4). Thus, the recommended modified flow regime is envisaged as encompassing a framework of commonly-occurring low flows, interspersed with selected higher flows each of which adheres to natural limits of magnitude, duration and timing, and performs certain functions.

The researchers say that the Kruger National Park Rivers Research Programme is greatly contributing to a more comprehensive approach to assessment of the water requirements of rivers(4).

In the past the main emphasis has been on the impoundment of water, as well as the reallocation of flow, or inter basin transfers. 'To date research has been dominated by ecologists, ... little input from chemists, botanists, geomorphologists and hydrologists" says Walmsley (1989) in an article : Water for Environmental Management - An overview. In late 1994, a announcement of funding of research on Water for Environmental Management by the FRD was made.

Following the interest in river quantity and not quality, research on riparian zones, riverine forests, wetlands etc. has been initiated by different organisations.

It must be appreciated that physical and chemical relationships of flowing waters, acting in concert with physiological, morphological and behavioural traits of lotic species, have resulted in a mosaic of complex community relationships that are easily disrupted by man's activities in controlling the quantity and seasonal distribution of flows(7). If a river is disrupted sufficiently, its self-purifying properties may be irreversibly impaired.

Undoubtedly, hydrological, geological, and topographical attributes form the primary driving forces for lotic ecosystems. Dependant on flow volume, rate and timing are many other major features of flowing systems: channel morphology, depth, hardness and stability of the diversity, extent of riparian zone, and flood plain morphology, groundwater turnover and quality etc. River diversion planning should take these factors into account to minimise impacts.

Riparian vegetation plays a direct role in the functioning of river systems through the effects it has on water quality and flow rate(8). It is for this reason that precaution should be taken when instream artificial wetland construction is being considered. The channel of flow of rivers might be significantly altered, and specialist advice may be necessary when considering the implementation.

The characteristics of the riparian zone depend on the geomorphology of the valley and the nature of the substratum over which the watercourse runs. Where the slopes are flat and the soils have a high water holding capacity the riparian zone will be wide, such as flood plains and permanent marsh.

The ecological development and significance of the riverine strip is usually related to stream order, being more developed and biologically diverse along mature rivers (stream order >5).

5. RIVER CHANNEL MODIFICATION DURING RIVER DIVERSIONS

Usually during river diversions, the flow channel is altered drastically. This results not only in different flow character, unnatural substrates and erosion of banks, but the riparian zone is reduced, leading to more artificial flow conditions and a reduced self-purifying capability(8).

Numerous studies have shown the importance of geomorphology in anticipating hydraulic and ecological stability in river channels(11). Geomorphological response to changing flow regimes of the Sabie and Letaba River systems is being studied in a WRC project to be finalised in 1995(10). The form of a river channel, which defines the physical environment for aquatic biota, is determined by water and sediment supply from upstream. Changes in these supplies, by for instance river diversions, result in significant modification of the river form and habitat.

Bruwer & Ashton (1989) divide human disruptions of river flow into:

- Impoundments (large dams)
- Weirs (walls < 5m high)
- Farm dams (< 250 000 m^3)
- Flood control facilities
- Off-channel storage facilities
- Hydropower facilities
- Pumped storage facilities
- Inter basin transfers
- Canalisation and bank stabilisation
- Direct abstraction
- Return flows(7).

Channel modification for various purposes has been both widespread and intensive in Wales and England. Streams have often been aligned to run parallel with straight fence lines, larger water courses were diverted or impounded to feed artificial lakes or to power mills. The Land Drainage Improvement Works Regulation (1988) and the Water Act has led to allowing the recovery, or facilitating restoration, of morphological characteristics to river channels that have already been modified. Rectifying past mistakes includes recreating pools and riffles, narrowing over widened channels, and removing culverts. The recent restoration project carried out in the river Thames catchment is outlined to highlight the use of geomorphological design criteria.

Given the fact that very few straightened river channels recover naturally, there is considerable scope for restoration. Reinstatement of bends is now common practise along straightened watercourses in Denmark(11).

Cut-off trenches and diversions on mines tend to be straight, and for these reasons the following factors need to be kept in mind.

Straightening river channels, decreases stream resistance through bends, increasing stream velocity and therefor erosion. If the sediment yield is limited, typically unstable sand or gravel bars form, and pool and riffle formation is restricted(11). Meander removal by stream alteration leads to shortened water retention time, and an increase in hydraulic energy dissipation. More energy is therefore available for bank erosion and sediment transport. Meander re-growth may occur only very rarely. Increased sediment transport results in increased channel instability, bed degradation, changed channel pattern, and sedimentation of downstream ponds and reservoirs. Biological degradation then sets in due to habitat loss. Not only is the substratum texture altered, but loss of riffles and pools result in reducing habitat for macro invertebrates and both habitat and spawning ground for fish. A loss in discharge buffering capacity is also experienced with increased flood peak flows.

Reduction of riparian vegetation through channel diversions leads to increases in sediment yield and solar radiation, to such an extent that macrophyte weed cutting in streams to reduce floods is now common practise in Denmark and England(12).

In Sweden a building block model for stream restoration has been developed. From this model specific restoration procedures can be used together or left out depending on the need of the stream and surrounding area. The stream restoration is not only being performed to restore plant and animal life but also to reduce agricultural nutrient loading(12). The building blocks include; buffer strip (interface between channel and terrestrial landscape) increased by riparian wetlands, revegetation of buffer strip with indigenous shrubs and trees, horseshoe wetland construction, side-slope reduction, meander valley formation enhancement, riffle/pool creation and riparian wetland/swamp forest construction.

Four key impacts of straightening river channels are addressed in this manner, namely; loss of stream length, loss of ripples and pools, loss of riparian flood plains and wetlands and loss of vegetation along the stream. All four these physical alterations change both the hydraulics and biology of the stream.

6. SEDIMENT DEPOSITION DUE TO RIVER DIVERSIONS

A factor that must be kept in mind is that erosion is often prevented or controlled, but little attention is given to deposition, which is just as important. This is specifically true for marshy wetland areas where stream velocity and carrying capacity is usually reduced.

Channelling of flow for mining purposes often ends abruptly, causing small scaled deposition deltas, which reduce flow and increases infiltration rates. The sediment can be removed if it is in a fairly open area, but if deposition takes place in a wetland area, removal of sediment or sludge often results in the entire wetland ecosystem being removed with it. In one occurrence, during a case study, for setting up guidelines for the assessment and amelioration of gold mining activities on the surface water environment[22], a metallurgical plant spill into a return water dam, resulted in expensive retrieval costs of sludge from wetland vegetation, and problems in processing the recovered sludge due to the vegetation matter in it.

The event occurred at a lined canal leading into a wetland, as is often the case on mines. Because wetland vegetation like *Thypha, Phragmites* and *Arundo* grows in shallow water, a preventative action for mines could be, to make sure that in the area where canals enter dams or rivers, a sediment trap within the dam or stream is formed. In this way, even if there is no wetland vegetation and an accidental spill occurs, the sediment/sludge can be removed from the sediment trap at a lower cost than when it is spread out on the bottom of the entire dam or wetland area.

By hollowing out the canal discharge areas, wetland species can still grow in remaining sections of the dams or rivers and perform the task of biological filtration and heavy metal accumulation.

Knighton (1991) from the Dept. of Geography (University of Sheffied), published some research on the influence of mining on river channel alterations. He found that in Tasmania, forty million cubic meters and one million cubic meters of mining waste were estimated to have been supplied respectively to the Ringarooma and George Rivers. Given the volumes involved and the fact that much of the input was less than 5 mm in diameter, the size composition of the bed material changed from gravel to sand as the rivers progressively aggraded their beds down stream. Where degradation has been occurring long enough, the bed material has again become gravely through re-exposure of the original bed and/or lag concentrations of coarser fractions within the introduced load. Such armouring should improve bed stability and slow the rate of degradation. Flow condition can also be altered by mines through discharges of water, increasing erosion and altering the channel bed sediment character, and also their planimetric and cross-section geometry[16].

Knighton (1991) also discusses river channel responses to mining waste. Mining waste introduced into a natural river can either be transported alongside the indigenous load without causing much disruption or result in a complete metamorphosis of the channel, two ends of a spectrum of response termed respectively "passive dispersal" and "active transformation". Initial braiding of stream flow is followed by an episodic incision process reflected in the development of terraces and associated narrowing of channels as it reverts to a single-thread form[16].

If canals have to be constructed, sediment deposition in the canals themselves may be reduced by using rectangular shaped canals (with the long side horizontal) as described in the work of Paul & Sakhuja[13]. Miller (1985) goes as far as showing examples of car wrecks being used to reduce erosion of berms in canalised rivers in the USA, but states that this is a drastic measure with aesthetically unacceptable consequences.

Blench (1972), gives very important methods for Application of Morphometric Relations. Not only can channel straightening and reservoir (impoundment) impacts be predicted, but specifically river diversion influences on representative discharge from meandering, braided streams, exits from lakes and sandy beds can be plotted. Channel width, depth and velocity of flow can also be predicted in terms of annual discharge. River-bed sand particle distribution alterations can also be predicted in terms of percentage by volume coarser than size ratio(18).

7. SUMMARY

Graphical plotting will show a power function relation of stream discharge, to mean water velocity, mean depth and width, slope, channel roughness, and suspended sediment(19). Alterations of any of these variables through river diversions, will lead to a change in the others. As river diversions inevitably lead to at least an alteration in discharge or channel width/depth, or channel roughness, most of the variables are expected to be affected.

In collaboration and consultation with expertise in this field, at least the morphology of the natural channel can be simulated, reducing impacts considerably.

On a national scale, the geomorphology of river channels as dominant factor for habitat and flow regimes should be studied, and a database concerning sensitive geomorphological river habitats could be drawn up using a Geographic Information System with its relational database facilities.

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INFORMATION MANAGEMENT

1. INTRODUCTION

In most industries, large amounts of data, that used to be stored in filing cabinets manually, are now being stored and managed electronically by computer. The advantages are becoming so obvious that it is almost considered counter productive not to manage growing data sets in computerised databases.

2. DATA MANAGEMENT

Improving hardware and software in the personal computer field, has put electronic file and data management within financial reach of most operations concerned with water quality management.

Storing water quality data in an electronic format, probably costs less than manually filing paper hard copies, certainly takes up less space and is found, edited and manipulated and exchanged much easier.

If a laboratory responsible for water sample analysis does not supply the data in an electronic format, and it needs to be entered into a computer software package, there should be a verification system whereby the integrity of the data can be checked. An ionic balance, as discussed in the literature review on water sampling and analysis, is usually useful in this regard.

For mines, the manipulation of data in electronic format can be of specific use for regular reports and water balances, that need to be submitted to the DWA&F as well as for use in EMPR's.

Data and information management need not be complicated to be effective. Large amounts of data can be stored, manipulated, edited, updated, queried and graphically displayed with inexpensive and user friendly spreadsheets (Lotus, Quatro-Pro, Excel, Works etc.) or database packages (Dbase, Oricale, etc.).

For the purpose of this report, data and information will be seen as synonymous and databases will be discussed as specific sets of data already stored in a spreadsheet or database package.

The aim of database management is to maintain data in a format in which it can be extracted or manipulated as quickly and as easily as possible. Keeping specific variables in individual rows (or records) and columns (items) is almost always the default option in most computer packages, and is becoming very important in open systems (to be discussed later). Vertical columns (items) are becoming the order of the day (Lotus, Excel, etc.) even though some databases do still require variables to be fed in, in continues strings, divided by commas or markers. The column/row data sets are more logical and can directly be imported into other packages (Word processing, statistical packages and Geographic Information Systems).

3. GEOGRAPHIC INFORMATION SYSTEMS (GIS)

Geographic information systems (GIS) are computer packages with the ability to link most databases with geographical or spatial data. GIS is a technology, that has moved, in less than a decade, from a 'background' technology used by specialist departments, to a management tool with limitations imposed only by our own lack of initiative(3). This makes the data more powerful as queries about the data can now be answered in the form of tables, graphs or maps, e.g. show the areas where: geology = shale, permeability < 0,01 cm/s, vegetation = grassland, slope < 10 degrees, distance to closest surface water > 2 kilometres. Answers to queries like these can be displayed as maps.

The characteristic that differentiates a GIS (Arc-Info, ILWIS, IDRISI, Map-info, Gene map, Regis, Intergraph, etc.) from general computer mapping or drawing systems (AutoCad, TurboCad, Powerpoint, Harvard Graphics, Draw perfect, etc.) is the link to the information/data base. Data can be attributed to a specific point, line or defined area, that has actual relation to surrounding features and an international grid or degree system. Once the database is constructed, correlation's between different pieces of information can be examined easily through computer-generated overlay maps.

The added advantage of location, distance between phenomena and area, almost leads to artificial intelligence that makes the data set an asset. The GIS can actively play a role in decision making, in this manner. Furthermore, data that had to be collected for one reason e.g. geology of an area, can later be used repeatedly in spatial manipulations and decision making. The pay off comes from the multiple ways in which the data can be used once it is digitally accessible in a GIS(8).

Furthermore, scenarios can be modelled through "what if" queries of the data base(6), e.g. permeability of a catchment, vegetation, geology, topography, rainfall, etc. is already available in a data set, now rainfall is increased to simulate a 1:100 year storm event - flooding can be simulated(1) in this manner to aid in long term planning.

Geographic information has been used in hydrological modelling, for many years but recent developments in the capabilities and speed in computer hardware, together with appropriate software applications, have greatly facilitated the manipulation, processing and display of geographic information using GIS. The Mgeni catchment study, is an example of a hydrological modelling coupled with GIS, that increased ease of data access, extraction and speed and efficiency of data manipulation(10).

For hydrological modelling purposes, there is generally an extra step of generating hydrological parameters that are dependent on data base information. This hydrology-GIS link is a significant factor because it involves empirical or physics-based relations. Applications of GIS in this field range from, synthesis and characterisation of hydrologic tendencies, to prediction of response to hydrologic events(8).

The prediction of movement of water contaminants is one of the applications most amenable to GIS. The use of GIS in urban water quality modelling, ground water contamination source delineation and discharge permit tracking is also becoming more applicable(8).

Integrated approaches, coupling water quality computer simulation modelling with GIS, have been used to delineate critical areas of non-point source pollution. On the basis of selected criteria for soil erosion rate, sediment yield and phosphorous (P) loading, areas with specific pollution potential can be identified(9). Complex, large area, multi-basin drainage studies require significant efforts in terms of data organisation, development and calibration of model parameters, and presentation of results. GIS provides the technology for these and less complicated studies that also need data management(7).

Because so many GIS applications exist for water management, it is not surprising that the largest GIS user in South Africa is the Department of Water Affairs. The departments' 10 year project concerning water resources and quality has lead to an extensive and expanding data base. The data base is available to all users in the Department on a LAN (Local Areas Network) and plans are underway to provide access to regional offices through the use of a WAN (Wide Area Network). More than 100 projects are underway in which data is being added to this GIS database(2).

"A GIS is a water manager's dream. To improve our understanding of the water situation in South Africa, we have introduced one at Water Affairs, and exciting developments are now taking place" says Mr A. Conley, Chief Engineer of the Directorate of Strategic Planning at Water Affairs(15). GIS assists, by digesting masses of data, and presenting implications of current situations, or alternative choices, as simply as possible. Conventional information systems were not solving the problems because they do not have the relational ability (topology) that a GIS has. Furthermore, GIS maps can be made to represent a library of information, or datasets. All data bases need to be kept current and up to date, to have relevant maps available with no time delays. Another aspect that favours GIS databases, is that when two databases are combined, one can create, and act on, new information that did not exist independently(15).

IT Southern with Southern Water Services, have, over 10 years, implemented a regional, 50 workstation fully networked GIS, throughout the South East of England; this now supports the operational activities of Southern's water business. The rationale for moving toward integrated, geographically based, information systems for effective management of assets is well illustrated, in this case.

3.1 INFORMATION MANAGEMENT FOR DECISION MAKING

Modern asset management does not just imply more information; it demands pertinent information. " Managers now have less time to make decisions, and need fast, accurate, information to make fast, accurate, decisions. Information configured in a spatial context can greatly assist this process(13)."

Meeting new performance levels demanded by customers and government regulatory mechanisms, compound the challenge, especially when " audit-proof " reporting is required, and when the whole information process must be cost effective against commercial objectives.

In South Africa, many projects are funded purely to capture data in spatial (geographic) data bases. If this is the case, as in any other projects, accuracy and currency is of the essence. There are warnings against costly time consuming data capture, this can be the most expensive and time consuming stage in the implementation and use of GIS. Once a data base is available and usable, the information system becomes cost effective. " Turn-around time from data capture to map analysis should be a matter of days, not weeks(19)." One solution is buying captured information for initial projects. In this manner a GIS can pay for itself sooner, as decisions and final products can initially be reached without the time consuming capture of data. As mentioned, the DWA&F has one of the largest GIS systems in the country, and large amounts of data are available already. Furthermore there are institutions like the CSIR, universities, municipalities and private companies that have considerable data bases available.

An international library, of GIS data sets, has also been established, in which Arc/Info (GIS) data concerning; country boundaries, political divisions, lakes and rivers, rail roads and roads, demographics, health, food production, labour force, education, economic, industrial and natural resource information is available(4).

In South Africa a national Environmental Potential Atlas (ENPAT-National) has been prepared on 25 data sets. The Atlas is a set of GIS digital maps (linked to the data sets), of the country, from which queries can be made. Five regional atlases of Durban, Gauteng, Cape Town, PE and Bloemfontein will also be released on a 1:50 000 scale - by the Dept. of Environmental Affairs(5).

Other users of GIS include, Johannesburg metropolitan planning, that have water and electricity supply facilities on a GIS and the Rand Water Board which also uses GIS in planning and service provision. Roodepoort municipality has been using a Sicad TDB (technical Data Base) since 1985. The water network component and associated data of the municipality is available in an " electronic inventory " in this way(17). Universities and private companies also have specific databases available.

Besides the statistics, graphics, spatial (mapping) and modelling capabilities that GIS makes possible for data sets, raster GIS packages usually have the added advantage of handling remotely sensed data (image processing).

Satellite images, aerial photographs and other remote sensing techniques are becoming a very important method of data acquisition for GIS databases . This data, however, needs to be altered and corrected from distortions before use, and most GIS packages (Ilwis, Regis, Idrisi etc.) have this capability. Not only are these images a source of new information, but it can contribute to modelling and database queries, because existing maps of land use, infrastructure, borders and boundaries etc. can be overlaid on these satellite images.

4. REMOTE SENSING

Remote sensing as a tool is becoming crucial in water and environmental management, and the added advantages of using remote sensing as a data capturing technique in a GIS, is becoming increasingly more cost effective.

In South Africa, ground water is becoming a very important resource. As solid waste disposal's impact on this resource is being felt more, the use of GIS and remote sensing in this field is increasing.

Mapping of aquifer susceptibility to pollution is considered a critical first step in implementing ground water management programs. GIS's have been used in many aspects of ground water management as well as modelling. Spatial models designed to evaluate ground water vulnerability to contamination have been more widely implemented in GIS than any other single type of ground water related model(21).

One such example is Erdas and ARC-INFO software being used with DRASTIC - a ground water modelling package. The acronym is derived from seven factors considered in the model:

Depth of water table, Recharge, Aquifer geology, Soil texture, Topography, Impact on vadose zone and Conductivity (hydraulic) of aquifer(21).

Modelling and monitoring with a combination of remote sensing, GIS and hydrological models, are useful in the management of this resource.

Evaluations of a low-cost, non-metric aerial mapping system for waste site investigations has proved the system useful and cost effective. Photos taken with a hand-held, standard 35-mm camera are enlarged using a commercial colour copier. Stereoscopic measurements are captured with a digitising tablet using conventional photogrammetric solutions. These aerial surveys were found to be well suited for waste site investigations as they balance accuracy with economy(19).

Virtually all waste site investigations require time- and geo-referenced information. Time referencing is important because chemical and biological interactions are related to cause and effect events. In addition investigators require data frequently, and often with short notice. Spatial location is important because many chemical and ecological reactions are influenced by topography(19). Another advantage was that, excluding the time it took to commercially develop the film, the entire process of, surveying control points, photographing the site, and issuing the map, took less than 24 working hours.

Remote sensing techniques of infrared thermography and ground penetrating radar can be used in combination to detect buried waste sites, buried tanks/pits, and both potentially hazardous and non hazardous fluid leak plumes(18).

Another use of remote sensing in waste management, in order to limit ground water contamination, is characterisation of waste facilities. Geological remote sensing can be used to characterise solid waste landfills and hazard waste sites in two ways: the use of digitised stereo photos to compute digital elevation models (DEM) of the landfill and the use of multi-spectral image data to map chemical compositional differences that are environmentally significant. The former can be used in water run-off prediction. The latter can likely be used to map clays, ferric oxides (Landsat band 5 & 4 (0,5 - 0,7 microns)) associated with contaminated ground water geochemical 'cells , stressed vegetation, and gaseous plumes escaping from landfill. (22)

Ferric oxides and hydroxides (such as hematite, limonite and goethite) can be mapped with multi-spectral scanners and are often found where water has discharged through gravel and sand. Under some circumstances the oxidation-reduction phenomena associated with contaminant leakage into ground water will deposit iron oxides in a manner similar to ferric and ferrous oxide exposures, related to geochemical cells, related to mineral exploration. This is an indirect approach similar to that used by exploration geologists, who map clay and sulphate exposures when looking for gold deposits that contain gold in quantities too small to detect directly.

It is also likely that surface exposures of exotic compounds with unique spectral features, such as cyanide and chromium, will result from leachate or contaminant interactions with soils and water. An example is "blue soil" associated with cyanide compounds, the blue soils can clearly be distinguished on a satellite image.

Historically, locating and browsing satellite data has been a cumbersome and expensive process. This has impeded the efficient and effective use of satellite data in the geosciences. SSABLE is a new interactive tool for the archive, browse, order, and distribution of satellite data. SSABLE has been evaluated at over 100 international sites. Network response time in the USA and Canada varies from 4 - 7 seconds for browse image updates and 20 - 25 seconds in Europe and Australia (20).

The last two decades has seen the geological sciences branch out into more and more specialised sub disciplines. Modern computer graphics technology and interactive data base handling heralds a time for reintegration. Geographical information systems are ideally suited to integrating large quantities of disparate data in a spatially intelligent fashion. In the geological sciences where data sets are becoming more unwieldy and complex, this characteristic is unparalleled as a solution[14].

Recently stylolite densities were cross-tabulated with gold concentrations and various ratios of trace elements. Spatial relationships among these variables were found which revealed new insights into the origin of gold deposits, and which can serve as a useful exploration axiom[14].

Satellite and other remote sensing methods are actively used in South African earth science applications. These include lithological and geochemical mapping, identification of fractures and lineaments, regional tectonic and geomorphic analysis, and various environmental applications. It has been found that this method of regional geological mapping is substantially more rapid and cost effective than traditional techniques[14].

For these and other reasons Gold Fields and the Chamber of Mines have contributed very generously towards establishing an Interactive Geographical Computer Centre at UCT. The Geology honours at RAU will include a GIS course in future, and the geology department at UP is also active in the GIS field.

Minescape, is an example of a software system that integrates the full spectrum of mine exploration and operating data into a three-dimensional graphic database. This capability, combined with the user's ability to directly view and manipulate data graphically, provides a realistic computer model for 3-D mining block generation, mining design and spoil dump planning. This 3-D geological model allows for such vital information as the presence of ground water reservoirs[21].

The role of GIS in Integrated Environmental Management is becoming as important as in water management. In a survey of 68 companies in 1989, 17 companies were using GIS and another 22 intended using GIS. The role of GIS and IEM in the landscape planning of catchments is also emphasised in specific WRC reports[16]. As mentioned earlier, GIS provides the technology for complicated studies of complex, large area, multi-basin drainage studies[7].

4.1 GIS APPLICATIONS IN THE ENGINEERING FIELD

In an article in The Civil Engineer,[11] the question is raised whether GIS is useful to the engineer.

Classically the engineer has written his own software, using his own database design (suitable for his own application). GIS packages should (and some do) therefor allow some additional programming facilities for customising. The type of queries/analysis performed by an engineer generally require additional software. This use of additional software (Basic, C, LISP, Pascal, SQL) is referred to as Third Party Development. It is this ability of GIS to cater for this type of third party development that makes it a really useful tool for the engineer.

5. OPEN SYSTEMS

The capability of GIS to accommodate additional software makes it the backbone of open systems which are probably the future in information technology. Though hardware independence is a necessity for long term data base planning, dynamic data management facilities that GIS provides, will become very important in future data management considerations.

MS-DOS may be replaced by OS/2 in time (11) (more likely WINDOWS NT), as OS/2 and UNIX are the most commonly used operating systems in the SA networking environment(11). A GIS system that can run on all three (or more) of these operating systems, will be ideal. " The multi-user capabilities of the OS/2 operating system make it ideal for database applications. " Existing PC's using DOS can be upgraded, at a nominal fee, to fit into the network. Regardless of the operating system, data should be kept in a format that can be imported into the most packages possible (but definitely including a GIS package).

Principally, data should be stored in an object-orientated GIS data structure. It must be kept in mind that the end product of a GIS is not only a drawing or map, but a live database that can be dynamically maintained at a variety of levels(11).

Examples of designed systems are discussed in some literature. One such system has been developed to provide a solution for the management and engineering analysis and design of water supply systems. The software is integrated with a graphical package, a database and a geographical existing database. The integrated approach provides a powerful engineering tool and a management system in terms of a database, data capture facilities and report generation(12). Many integrated systems, like these, are being developed, and most systems have at least the ability to be linked with a geographic data base or a GIS directly.

The future of information technology is probably best illustrated in an European example of how an open system was implemented in a municipal water supply division. An engineering software system with electronic detectors on supply pipelines was implemented to detect pressure changes due to leaks or bursts. The engineering system is linked to a GIS which gives each electronic detector a spatial location, in the database. The GIS is also attached to the municipal database (containing water account information, streets, addresses, pipelines etc.), so that in the case of a pipe bursting, it is detected by the detectors of the engineering system, the location and affected areas are isolated by the GIS, which causes the accounts to be adjusted according to the time that the user was without water. Apart from the obvious advantage of such a system to the rate payer, amounts of water lost by leaks and bursts are limited, and the municipality is approaching an intelligent open system. The possibility of applying this scenario to large water using industries, like the mining industry, in South Africa, should be researched thoroughly.

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