CHAPTER 2

LITERATURE REVIEW

2.1 Introduction

In open cast and underground coal mining operations, large volumes of coal-mine water from aquifers are released inadvertently through coalfaces. The coalfaces contain groups of minerals of metallic sulphides called pyrites, which can easily create sulphuric acid (acid mine water) when they come into contact with the released aquifer water (Gladney *et al*., 1983). This water is a major problem for coal-mines throughout the world (Kupchella and Hyland, 1993). The devastating effect of such waters is associated with its acidity (between pH 2 - 4). Some coal-mines also generate water qualities associated with calcium, magnesium, sodium, sulphate, carbonate and bicarbonate, with near neutral pH depending on the geology of the area.

Various studies show that some of these waters cannot be used for domestic purposes and/or released into natural streams, unless some form of water treatment is applied to nullify or neutralize its acid levels. Liming plants are usually used to treat the water and reduce its acidity levels to between pH 5 and 9.5 before the water can be utilized. However, the cost of running such liming plants is very high and thus, alternative methods have been sought. In most cases, after neutralization, the water is too saline to release to streams. These neutral mine waters need additional treatment, unless they can be utilized through some other technology, like irrigation of agricultural crops (Annandale *et al*., 1999).

A survey of literature reveals that agricultural use of mine water *per se* is limited; however, several reports are available on saline and/or sodic water use for irrigation of agricultural crops. Mine water is often very poor in quality, and can thus be classified as saline and/or sodic waters. The available literature on saline and/or sodic water, therefore, can be applied to the concept of coal-mine water irrigation, and in this literature review, the local (South African) and international knowledge available on irrigation with saline and/or sodic water is considered.

2.2 Soil and crop response to saline and saline-sodic water

Soils and crops respond to saline and saline-sodic irrigation either positively or negatively, depending on the composition and salt concentration of the water. For successful use of saline water for agriculture, therefore, selection of salt tolerant crops, suitable irrigation management strategies and the choice of appropriate irrigation systems is essential (Rhoades and Loveday, 1990).

2.2.1 Crop response to salinity

There are two ways in which saline waters affect plant growth: (1) when salts in the irrigation water decrease the osmotic potential of soil water and (2) when ions in the soil water exceeds a certain concentration value and become toxic to plants.

Effect of salinity on osmotic potential

Plants extract water from the soil when leaf water potential is less than total soil water potential. Total soil water potential is the sum of matric, osmotic and gravitational potential of the soil water. Salinity affects plant growth by decreasing the osmotic potential of the soil water. Plants close stomata when water is unavailable as a result of decreased osmotic potential in the soil water. Depending on the plant species, stomata begin to close when leaf water potential reaches -500 to -1500 kPa (Boyer, 1974), which leads to a reduction in photosynthesis. When leaf water potential reaches -1500 to -3000 kPa, the stomata are completely closed and photosynthesis ceases (Begg & Turner, 1976). Leaf enlargement and other growth processes begin to be affected at even higher (less negative) leaf water potential values than those which affect photosynthesis (Boyer, 1970 & Hsiao, 1973). According to Boyer (1974), plant growth may be reduced even if matric potential is close to zero, if the concentration of soluble salts in the soil water is high enough to lower osmotic potential to several hundred negative kPa. A matric potential close to zero implies that the soil water content is high. This indicates that a high salt concentration has the same impact on plant growth as low soil water content, the latter being associated with a low matric potential.

Toxicity

The concentration at which toxicity affects plant growth depends on the ion and plant species involved (Bernstein *et al*., 1974). Ions like boron, sodium and chloride in irrigation water can cause toxicity in certain crops. Ayers and Westcot (1985) present recommended maximum concentrations of trace elements in irrigation water. Specific ion effects may involve direct toxicity or nutritional imbalance (Berstein & Haward, 1958; Orcutt & Nilsen, 2000). The detrimental effects of ions can be observed at the level of enzyme activity, membrane function and several important metabolic process, including photosynthesis and respiration (Orcutt & Nilsen, 2000). Under saline conditions, which are characterized by low nutrient ion activities and extreme ratios of Na/Ca, Na/K, Ca/Mg and Cl/NO_3, nutritional disorders can develop and crop growth may be reduced. Nutrient imbalance may result from the effect of salinity on nutrient availability, competitive ion uptake, transport of or partitioning of ions within the plant, or may be caused by physiological inactivation of a given nutrient, resulting in an increase in the internal requirement for the essential element (Mengel & Kirkby, 1987). Excessive amounts of Na salts in soil water reduce Ca availability as well as transport and mobility of Ca to growing regions of the plant. Salinity can also directly affect ion uptake due to competition for uptake through cell membranes as Na decreases K, Cl and NO_3 uptake (Grattan & Grieve, 1994). Most of the works that has been done on toxicity are compiled in the hand books quoted here. Not much work has been done since then.

2.2.2 Soil salinity

As water is taken up by the crop or evaporates from the soil surface, salts are left behind and accumulate. Each plant has a maximum soil salinity level that it can tolerate without negatively influencing yield or crop quality due to osmotic and/or specific ion effects (Maas & Hoffman, 1977; Maas, 1987). The salts need to be leached below the root-zone and according to Ayers & Westcott (1985), the leaching requirement (LR) can be estimated as $LR= EC_{iw}/[5(EC_e-EC_{iw})]$ where EC_{iw} and EC_e refer to irrigation water salinity and the crop tolerance to soil salinity. LR is the amount of additional water to be applied in excess of crop water requirement to prevent salt accumulation. LR increases as the EC of the irrigation water increases. In addition, LR depends on the initial profile salt content of the soil, the required level of soil salinity after leaching, the depth to which leaching is required, and soil chemical and physical properties (Ayers & Westcott, 1985; Abrol *et al*., 1988; Hoffman & Durnford, 1999). It is, however, not necessary to achieve with every irrigation event and leaching is only needed once the levels of soil salinity approaches hazardous levels (Oster, 1994).

2.2.3 Soil sodicity

High salt concentration and toxic salt levels do not damage or affect the physical properties of a soil (Shainberg & Letey, 1984). Irrigation waters with high sodium levels, however, tend to produce soils with high exchangeable sodium levels. Such soils frequently crust, swell, disperse and decrease the infiltratrability. High Sodium Adsorption Ratio's (SAR) increase infiltration problems, but if the irrigation water also contains high levels of salinity, the infiltration hazard is lessened. Du Plessis & Shainberg (1985) carried out a study on infiltration rates of South African soils using a rainfall simulator and results confirmed that some soils are very susceptible to crust formation at exchangeable sodium percentages (ESP) as low as one. Sumner's (1993) study also showed that soils with very low levels of exchangeable sodium can exhibit sodic behaviour in the presence of low salinity water. Ayers & Westcot (1985) published guidelines to indicate the severity of expected infiltration problems based on SAR and EC of the irrigation water. A severe reduction in infiltration is likely to occur with the condition of relatively low EC and high SAR (Figure 2.1).

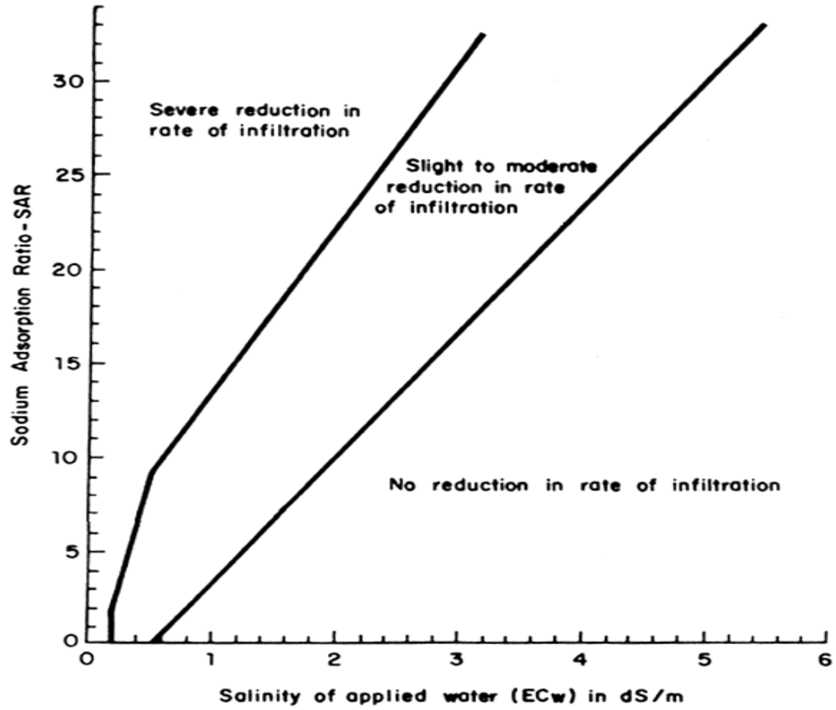

Figure 2.1 Potential for reduction in infiltration rates resulting from various combinations of EC and SAR of applied water (Ayers & Westcot, 1985)

Infiltration problems due to high SAR can be improved by adding gypsum to the soil or to the irrigation water. When the irrigation water comes into contact with gypsum, it dissolves into Ca and SO_4 ions that may slightly increase the salinity of the water, but simultaneously reducing the SAR. The Ca cations are then free to displace Na cations adsorbed onto the negatively charged clay particles, thereby enhancing flocculation, improving soil structure, and increasing the infiltratability.

The capacity of the SAR (SAR= Na/((Ca+Mg)/2)$^{\wedge 1/2}$) and ESP (ESP = (Na/ (Ca + Mg + Na + K))*100) equations to predict sodicity hazard from irrigation water quality and soil exchange sites is often complicated by evapotranspiration and changes in calcium solubility in the soil water that take place due to precipitation or dissolution (Ayers & Westcot, 1985). Shainberg & Letey (1984), and Rhoades & Loveday (1990), also noted that the change in the concentration of irrigation water and soil solution during a growing period are more important parameters than ESP for predicting the effect of sodicity hazard to the soil. Suarez (1981) introduced adjSAR to estimate the tendency of $CaCO_3$ to dissolve or precipitate, following irrigations and this parameter improved the capacity of SAR and ESP to predict soil physical problems. The permeability hazard, however, can be evaluated according to the relationship described by Rhoades & Loveday (1990) between adjSAR and the EC of the irrigation water.

2.3 Modelling the effects of saline-sodic water irrigation on crop growth

The need to assess sustainable use of coal-mine water for irrigation with regard to crop production, and the effect on soil chemical and physical properties, increased over the last decade (Annandale *et al*., 2007b). Such relations between irrigation water quality, crop growth, irrigation management and fertilization under different soil and cropping systems is complex, and needs well designed long-term field experiments. Long-term field experiments of such complex interactions, of course, are time consuming and expensive. Annandale *et al*. (2001) developed a soil water and salt balance model called SWB (Soil Water Balance) to manage irrigation with these water qualities and to provide insights into long-term effects of such waters on crop growth, soil water and the salt balance. The idea of this computer modelling study was also to assess the feasibility of using mine water for large scale irrigation, and predict the quantity and quality of irrigation return flows to groundwater and river systems. The model, however, would benefit from field-scale testing for a range of soil types, irrigation water qualities and cropping practices. In the following section, root zone modelling will be discussed. Return flows from mine water irrigated fields will be discussed in section 2.5.

2.3.1 Root zone modelling

The root zone is a dynamic region in a soil profile, with continual changes in water content, plant uptake of water and salts (Suarez, 2001). Water and solute movement, and root water uptake in this region are modelled in detail to accurately simulate the soil water and salt balance (Cardon & Letey, 1992b). There are several detailed root zone-salinity management models available in the literature. Clarke (1973) categorized such models into four groups: stochastic conceptual, stochastic empirical, deterministic conceptual and deterministic empirical. A model is considered as stochastic if any of the variables in its mathematical expression are described by a probability distribution. A model is termed deterministic if all variables are of from random variations. Models are conceptual if their functional form is derived from consideration of physical processes, and empirical if not. Addiscott & Wagenet (1985) also classified available models into deterministic and stochastic with the same definitions as that of Clarke (1973). SWB is a deterministic conceptual model.

The most recent review of model classification is by Hoffman *et al*. (1990), who classified them as transient and seasonal models. The seasonal models consist of equations that relate the amount of applied water to the seasonal yield, yield to average root zone salinity, yield to evapotranspiration (ET) and average root zone salinity to leaching fraction (LF) (Letey *et al*., 1985, Knapp, 1999). These models assume steady-state conditions and do not include crop response to variation in water content, weather, and soil salinity in space and time (Bresler, 1986). According to Bresler & Hoffman (1984) such models are not suitable for irrigation management under saline conditions. Examples of this type of model are WATSUIT (Rhoades, 1987) and SWAM (Singh *et al*., 1996). Research carried out by Letey *et al*. (1985) and Prendergast (1993) also report that these models may sometimes give results that could agree with observed field data, but have limited applications.

The transient models simulate water and solute movement in soil (Wagenet & Hutson, 1989; Cardon & Letey, 1992a). Water and solute movement in the soil, and root water uptake, are modelled in detail. However, the crop growth description is simple and does not consider interactions with environmental variables and agronomic management (Cardon & Letey, 1992b) an example of this is SWAP93 (Van Dam *et al*., 1997). According to Majeed *et al*., (1994), applications of such models for management of irrigation with saline water require a mechanistic description of relevant processes in the soil-water-plant-atmosphere continuum and proper interaction of these processes with crop growth. The Root Zone Water Quality Model (RZWQM) (RZWQ Team., 1998) and the SWB model are a few examples of mechanistic models in the USA and in southern Africa. Soil Water Balance (SWB) (Annandale *et al*., 1998) is a mechanistic, multi-layer, daily time step, soil water-salt balance-generic crop growth model, locally developed and parameterised for many crops.

Simunek *et al*. 2003 also recently reviewed various approaches for modeling preferential and non-equilibrium flow and transport in the vadose zone. The existing root zone water flow modelling approaches differ in terms of their underlying assumptions and complexity. They range from relatively simplistic models to more complex physically based models. According to Larsson and Jarvis (1999), the limited availability of comprehensive data sets has so far restricted the field validation of preferential flow models.

The applicability of the existing models to irrigation with mine water depends on the degree to which the models accurately represent the natural processes. For example, UNSATCHEM (Suarez, 2005) has unique features such as prediction of CO_2 concentration in the root zone, consideration of the effects of soil chemistry on hydraulic properties and inclusion of a kinetic model to describe the calcite dissolution and precipitation. SWB simulations have been found to be satisfactory for gypsum precipitation when compared to the out puts of UNSATCHEM.

2.3.2 Application of root zone modelling

Models have been used extensively to simulate field conditions for understanding basic processes and the long-term effects of various management options on the soil water and salt balance at field scale (Annandale *et al*., 2007a; Gates *et al*., 2002; Sarwar & Bastiaanssen, 2001). Particularly, validated mechanistic models have some advantages over long-term field experiments with respect to synthesizing information inexpensively and quickly. However, the reliability of model results is contingent upon the degree to which the models accurately represent the natural processes. Thus, model results must be compared to results from field experiments to ascertain the degree of model performance.

In most root zone model applications, the model is calibrated using a single season's experimental results and then evaluated with data from other years. This type of evaluation may not be effective if weather conditions are similar in all the study years. Another technique is to calibrate the model in one location and evaluate it in another location. Preferably, model evaluation should cover a broad range of management effects and locations. Good model predictions depend on model input parameters and model concepts as well as representative experimental data (Singh, *et al*., 1996). Evaluation of a model can only be objective if model users can give representative model input parameters. Some model parameters cannot be measured in one single experiment; therefore calibration of certain parameters is possible to achieve desired output (Donigian *et al*., 1995).

As noted above, model calibration and validation are necessary and critical steps in any root zone model application. Model performance and calibration/validation are evaluated through qualitative and quantitative measures, involving both graphical comparisons and statistical tests (Donigian, 1995). Comparisons of simulated and observed variables should be performed for daily, monthly, and annual values. Statistical procedures can include error statistics, correlation and model-fit efficiency coefficients, and goodness-of-fit tests.

2.3.3 Field scale application of the SWB model

The theory, classification and validation of root zone modelling approaches in general have been discussed. In this section, the SWB model is considered as an example of a root zone model that has been widely applied to field conditions in the southern Africa.

Model description

Soil Water Balance (SWB) is a mechanistic, multi-layer, daily time step, soil water-salt balance-generic crop growth model, developed from NEWSWB, a modified version of the model published by Campbell & Diaz (1988).

The first components of the soil water balance, which are calculated on a daily time step, are canopy interception of water and surface runoff. Water infiltration and redistribution can then be calculated using either a cascading soil water balance or a finite-difference water movement module based on Richards' equation. In the case of the cascading water balance, salt redistribution is determined assuming complete mixing of irrigation and rainfall with the soil solution of the topsoil layer, and similarly for the solution percolating to the next lower soil layer and so on. Any water that passes beyond the bottom layer is assumed lost to deep percolation. The amount of salt leached is then calculated from the amount and quality of the drained water.

Chemical equilibrium is calculated on a daily time step per soil layer, using the model published by Robbins (1991). The model of Robbins (1991) solves chemical equilibrium by iteration. Within each iteration, activity coefficients and ion activities are calculated for Ca, Mg, Na, H, SO_4, HCO_3 and CO_3, and the solution phase is equilibrated with solid phase lime and gypsum, if present. EC is calculated from individual ion concentrations (McNeal *et al*., 1970) for each soil layer. The SWB model ends the iteration procedure when the change in EC between the previous and the following loop is < 0.01 mS m^{-1}.

Potential evapotranspiration (PET) is calculated as a function of daily average air temperature, vapour pressure deficit, radiation and wind speed, adopting the internationally standardized FAO Penman-Monteith methodology (Allen *et al*., 1998). The two components of PET (potential evaporation and potential transpiration) are estimated from canopy cover. Actual transpiration is determined on a daily basis as the lesser of root water uptake or maximum loss rate (supply or demand limited). Total soil water potential is used to determine the amount of water available for crop transpiration in each soil layer. The osmotic effect on crop growth is simulated by adding osmotic potential to the matric and gravitational soil water potentials. Osmotic potential is calculated as a function of ionic concentration (Campbell, 1985). The daily dry matter increment (DM_i) is taken as the minimum of the water supply limited (Tanner & Sinclair, 1983) and radiation limited DM_i (Monteith, 1977). A stress index, the ratio between actual and potential transpiration, is used as a limiting factor for canopy growth.

Required weather and management input data are planting date, latitude, altitude, rainfall and irrigation water amounts and quality, as well as maximum and minimum daily temperature. In the absence of measured data, SWB estimates solar radiation, vapour pressure and wind speed according to the FAO recommendations (Allen *et al*., 1998). Required soil input data are volumetric field capacity, permanent wilting point and a runoff curve number to calculate runoff based on the SCS method (Stewart *et al*., 1976). In addition, initial volumetric soil water content, the content of soluble and exchangeable ionic species, as well as initial gypsum and lime are required for each soil layer.

If cascading redistribution is used, a drainage factor (fraction of water above field capacity that cascades daily to the next layer) and a drainage rate upper limit (maximum amount of water that can percolate from the bottom layer in a day) needs to be entered. The SWB model is written in Delphi v. 7.0 (Inprise Corp.) and runs in a user-friendly Windows 95 environment. The SWB model includes a database of specific crop growth parameters for 137 species (Annandale *et al*., 2007b).

Model application

Jovanovic *et al*. (1998), used SWB to predict the soil water and salt balance of lime treated acid mine water irrigated crops. Simulations were done using calculated crop growth co-efficients fitted measured data of water balance and crop growth. The predictions of the crop growth, soil water content and soil solution EC_e for single season simulations gave good agreement with observed data. Annandale *et al*. (2001) recommended that the SWB model should be further refined and validated for a range of soil types, irrigation water qualities and cropping practices. Further improvements and refinements should also be made to the runoff subroutine of the model. Beletse *et al*. (2004) also validated SWB for pastures irrigated using sodium sulphate rich mine water. Results showed that crop growth, soil water content and soil solution EC_e were well simulated, and good agreement was found between observed and predicted values. SWB model output of return flow from mine water irrigated areas was used as input into a groundwater model and the authors concluded that the impact of irrigation with mine water on ground water was simulated quite well (Annandale *et al*., 2006).

2.4 Irrigation with mine water in southern Africa

South Africa is the leading country in terms of mining in the southern part of Africa and mining contributes about 8% to the economy of the country. Coal mining in South Africa, in particular, is a very important industry, with a total of 65 collieries operating throughout the country (Pulles *et al*., 1995), and is the largest foreign exchange earner after gold.

Many of South Africa's largest coal-mines are located within the Witbank Coalfields in the Mpumalanga Province (Jones & Wagner, 1997). These coalmines consist of both underground and opencast workings. A large amount of low water quality is generated from these coal-mining activities and is in excess for coal beneficiation, road wetting, slurry dams and other activities. Pulles, *et al*. 2001 investigated the over all water balance of the South African coal mining industry and indicated that on average 133 ℓ of water is used for each ton of coal that is mined. They also reported that on average a mine use 77 963 m^3 day^{-1} for coal beneficiation and 13 064 m^3 day^{-1} for road wetting.

Mine water is unsuitable for direct discharge to the river systems except in periods of high rainfall when an adequate dilution capacity is present and controlled release is permitted (Pulles *et al*., 1996). A number of alternative desalinization treatment technologies were investigated (van Zyl, *et al*., 2000) where treated mine water must meet more stringent quality requirements (eg. <200 mg ℓ^{-1}). The capital cost of this process varied between R4 million/M ℓ/d and R10 million/M ℓ/d and the running cost between R2/m^3 and R5/m^3.

South Africa is a dry country with an average annual rainfall of only 464 mm, compared with a world average of 860 mm (Scott *et al*., 1998). Sixty five percent of the country has an annual rainfall of less than 500 mm, usually regarded as the absolute minimum for successful summer season dry-land farming. For this reason the available marginal and low quality water resources, such as mine water generated during mining operations, are becoming under an increasingly important consideration for irrigation purposes.

South Africa is the first country to test mine water for irrigation of agricultural crops in the region. The possible utilization of mine water for irrigation of agricultural crops was first evaluated by Du Plessis in 1983. He observed that gypsum rich water would be more suitable for irrigation than NaCl water (other water of similar concentration but with other ions). Large amounts of wastewater could possibly be made available to the farming community and utilised for the irrigation of highly productive soils in the coalfields of the Mpumalanga Province in South Africa, where water resources for irrigation are already under extreme pressure (Annandale *et al*., 2007b). In Botswana, studies have been done to consider the effects of the use of mine wastewater for irrigation (Jovanovic *et al*., 2001). Government has reserved this right to use this plan in future (Rahm *et al*., 2006). However, investigations are, in general, ongoing regarding the feasibility of wastewater use in agriculture (Rahm *et al*., 2006).

2.4.1 Composition of mine water

Throughout coal mining operations (open cast and underground), large volumes of mine water are produced and the composition of the mine water depends on the geology of the area. The water produced could be highly acidic (acid mine drainage (AMD)), which is characterized by low pH (pH<4) and elevated concentrations of dissolved heavy metals (Johnson, 2000).

Mining companies commonly use lime to treat the AMD. The water that results after treatment is rich in $CaSO_4$, $MgSO_4$, or Na_2SO_4 and pH remains between 5.0 and 9.5. Neutral pH waters at high total dissolved salts rich in Ca, Mg, Na and SO4 are also produced. Example of this is indicated in Table 2.1.

Table 2.1 Average mine water quality for Witbank (Annandale *et al*., 1999)

Analyses	Field	
(mg ℓ^{-1})	Major and Jacuzzi	Tweefontein
Al	0.3	0.01
Ca	513	405
Mg	158	196
Na	51	47
Fe	0.3	0.08
Mn	6	0.01
SO_4	2027	1464
Cl	18	32
HCO_3	143	68
TDS	2917	2212
pH	6.4	7.0
EC (mS m^{-1})	294	205

The listed water chemistries in Table 2.1 reflect a typical analysis of mine water for Kleinkopjé Colliery. As can be noted from Table 2.1, the lower the pH of water, the greater the presence of dissolved salts is likely to be. This is attributed to the fact that the salts dissociate and go into solution at reduced pH values. This can also explain the high TDS value of the water.

2.4.2 Gypsum precipitation in a soil – the opportunity to remove salt from the soil water system

The concept of gypsum precipitation in a soil arose as an opportunity in the South African coal mining industry that reduces salt leaching when lime treated AMD water was first used for irrigation of agricultural purposes (Meiring, 1983). This concept, which is protecting the environment by precipitating gypsum from the soil water system could be an opportunity in reducing salt leaching. The mechanism is that when this water is irrigated to a soil, crops concentrate up the soil solution through water uptake, gypsum precipitates and changes into solid phase.

Several studies have been undertaken worldwide, on the precipitation and dissolution of gypsum in soil but few in soils irrigated with gypsiferous waters. The studies indicated that gypsum precipitates when it reaches its saturation index. This index shows the status of a solution phase and is quantified by comparing its ion activity product (IAP) to solubility product (K_{sp}) of the solid phase.

Numerous studies indicate that gypsum precipitation in a soil is controlled by Ca concentration, pH and saturation of CO_3 and HCO_3. A high amount of Ca, low pH and amount of CO_3 and HCO_3 in a soil water system lead to increased precipitation of gypsum. pH level 3-5 is favourable for gypsum precipitation, but pH $<$ 2 solubilizes gypsum. pH 3-5 also controls Ca desorption from a solid phase and, CO_3 and HCO_3 concentrations in the system.

The largest impact of this gypsum rich mine water on the environment could be salinization of water resources. Du Plessis (1983) evaluated, using a steady-state chemical equilibrium model (Oster & Rhoades, 1975), the amount of salt that would leach from a soil, and could potentially contaminate groundwater. In his study he was able to explain that when irrigating with gypsiferous water, soil salinity and percolate water salinity was lower compared to when a chloride rich water of otherwise similar ionic composition was used for irrigation. Using a field scale model, Annandale *et al*. (1999) predicted that low soil salinity and percolate salinity could be maintained by irrigating crops using gypsiferous water. Both studies indicated that percolate salinity could be reduced as a result of gypsum precipitation in the soil (Jovanovic *et al*., 2001)

Annandale *et al*. (2001) carried out a field trial and indicated that by irrigating with gypsiferous mine water, a large fraction of the salts can be removed from the soil water system through precipitation of gypsum in the soil profile, as the soil solution is concentrated by root water uptake. This could reduce the likelihood of off-site environmental pollution.

Annandale *et al*. (2002) also described that the use of gypsum-rich mine water for irrigation of agricultural crops was a 'simple technology' principle. Salt leaching is considered to be limited as (1) Ca and SO_4 ions precipitate out as gypsum (2) redissolution of gypsum is a slow process (3) even after redissolution, gypsum occurs as soluble Ca and SO_4 that rarely gets adsorbed to the ion exchange site once the base saturation of the exchange complex is reached, and can easily leach from the soil system.

2.4.3 Crop production using coal-mine water

Coal-mine water is usually saline water that can be of various compositions of $CaSO_4$, Na_2SO_4, $MgSO_4$ or $NaHCO_3$ and commonly is dominated by cations such as Ca, Mg and Na, as well as dominant anions such as SO_4, HCO_3 and Cl. Not much work has been done on the effect of mine waters on crop growth and soil properties (Annandale *et al*., 2001), several studies have been made of saline irrigation waters that mainly consist of NaCl as the salinization agent (Grattan & Grieve, 1999). A number of studies have also examined crop response under solutions of various anionic compositions, particularly SO_4 and HCO_3, in controlled conditions in glasshouses. The effect of SO_4 and HCO_3 on crop growth will be discussed in this section as the irrigation waters used in this study are predominantly $CaSO_4$, sodium sulphate (Na_2SO_4) or sodium bicarbonate ($NaHCO_3$) rich. In addition, Annandale *et al*. (2001) report possible nutritional problems, like for example deficiencies in K, Mg and NO_3, that can occur due to using mine water irrigation for irrigation. Therefore, a portion of the following section will focus on the effect of salinity on crop nutrition, specifically of N, K and Mg.

Irrigation with $CaSO_4$ water

Effect of sulphate on crop growth

The threshold sulphate concentration which most crops can tolerate is 4800 mg l^{-1}(Mengel & Kirkiby, 1987). Sulphate is not toxic to plants, but its effect on plant growth is related to the cation associated with the SO_4 ion. Sulphate affects the associated cation by causing an ionic effect, unavailability of nutrients and hindering mobility or transport of other nutrients. The ionic effect of SO_4 on Ca, for example, is to decrease the Ca concentration through precipitation. The availability of nutrients is then influenced by the formation of gypsum. For instance, in Na_2SO_4 rich systems Ca availability is reduced through formation of gypsum. SO_4 reduces the uptake of other ions such as Mo and NO_3 (Martinez & Cerdá *et al*., 1989). The tolerance of most crops to sulphate toxicity is prevented through a series of metabolic processes. It is therefore unlikely that excess sulphate would influence growth through ion toxicity (Rennenberg, 1984).

Crops such as maize, sorghum, pearl millet and Lucerne are more sensitive to $CaSO_4$ rich water in the seedling growth stage than crops where tolerance is mainly connected to ionic effects of Na and Cl. Mentz (2001) observed that crops which are tolerant to salinity, tolerated high SO_4 concentrations.

Soil irrigated using $CaSO_4$ rich mine water in South Africa (Du Plessis, 1983; Annandale *et al*., 1999, Annandale *et al*., 2001; Annandale *et al*., 2002, Jovanovic *et al*., 2002) stabilised at a relatively low EC_e. The EC oscillated at around 200 mS m^{-1}, which is typical for a saturated gypsum solution (Annandale *et al*., 1999; Jovanovic *et al*., 1998). Du Plessis (1983) also reported that irrigating with lime treated acid mine water did not pose a problem to soil physical properties. The use of high concentration $CaSO_4$ rich waters for irrigation of agricultural crops is believed to be beneficial for crop growth as salt build up is restricted by the low solubility and precipitation of gypsum. Gypsum precipitated in a soil provides calcium, which is needed to flocculate clays in acid and alkaline soils (Shainberg *et al*., 1989, Sumner 1993, Sumner and Miller 1992).

Irrigation with $NaSO_4$

Sodicity is one of the most important problems related with Na_2SO_4 water that limits crop productivity. Its effect is complicated by indirect means such as induced nutritional imbalances and impairment of soil physical conditions (Maas, 1987). The effect of Na containing waters on crop production is discussed in detail in section 2.2.1.

Irrigation with $NaHCO_3$ water

Effect of bicarbonate on crop growth

HCO_3 affects plant growth through a decrease in the solubility of nutrients. The decrease in solubility is caused by the increase of pH associated with increasing concentrations of carbonates (Grattan & Grieve, 1999). For example, the concentration of soluble Fe in soil decreases 1000 fold per unit increase in pH. Zinc, Cu, and Mn are also less soluble at alkalinity-induced high pH (Barber, 1995). The high pH caused by alkalinity may directly inhibit growth of sensitive plants, as demonstrated in Lupinus species (Tang & Robson, 1993). However, in most instances it is not the pH, but the high concentration of HCO_3 that is the major factor for plant growth inhibition (Lee & Woolhouse, 1969) due to its toxic effect. This was demonstrated by maintaining maize plants growing in solution at pH 8.0 with and without HCO_3. The high pH without high HCO_3 did not cause any negative effect on root and shoot elongation (Lee & Woolhouse, 1969).

Plants respond to elevated HCO_3 concentrations with decreased shoot growth. Shoot growth inhibition is associated with a decrease in number of leaves, fresh and dry mass, and shoot elongation. Sunflower (Alcántara *et al*., 2000), tomato, and petunia (Bailey & Hammer, 1986), chrysanthemum (Kramer & Peterson, 1990), apple (Zhou *et al*., 1984), rice (Yang *et al*., 1994), sorghum, maize and barley (Alhendawi *et al*., 1997), grapevine (Römheld, 2000), olive, peach (De LaGuardia & Alcántara, 2002), pea (Zribi & Gharsalli, 2002), and roses (Fernández-Falcón *et al*., 2006), exhibited stunted growth when growing in either soil or nutrient solution containing a high concentration of HCO_3. The detrimental concentration for HCO_3 reported varies between 4 and 20 mM.

Salinity effect on Nitrogen (N), Potassium (K) and Magnesium (Mg) availability

There is no clear evidence indicating that N applied to saline soils improves plant growth or yield. A number of laboratory and greenhouse studies have shown that salinity can reduce N accumulation in plants (Cram 1973; Pessarakli & Tucker, 1988; Feigin *et al*., 1991; Pessarakli, 1991; Al-Rawahy *et al*., 1992). Many attributed this reduction to Cl antagonism of NO_3 uptake (Bar *et al*., 1997; Feigin *et al*., 1987) while others attributed the response to salinity's effect on reduced water uptake (Lea-Cox & Syvertsen, 1993). The form in which N is supplied to salt-stressed plants can also influence salinity-N relations as well as affect salinity's relation with other nutrients (Lewis *et al*., 1989; Martinez & Cerdá, 1989). NH_4

supplied maize (Lewis *et al*., 1989), melon (Feigin, 1990) and pea, *Pisum sativum* L. (Speer *et al*., 1994) plants were found to be more sensitive to salinity than NO_3 supplied plants when grown in solution cultures.

According to Lewis *et al*. (1989), addition of Ca to growing media improved the growth rate of the plants in the NO_3 treatment, but not those treated with NH_4. Martinez and Cerdá (1989) also found that Cl uptake was reduced in cucumber when only NO_3 was added to the solution but when half the NO_3 in the solution was replaced by NH_4, Cl accumulation was enhanced. These investigators further noted that when NO_3 was the only N-source, accumulation of K in the plant was increased under saline conditions. As the NH_4/ NO_3 ratio was increased, plants accumulated more Na and Cl and less Ca and K in their leaves. Numerous other studies with a wide variety of crops have also shown that K concentration in plant tissue declines as the Na-salinity or as the Na/Ca ratio in the root media is increased (e.g. Francois, 1984; Graifenberg *et al*., 1995).

Most salinity-nutrition studies have given little attention to magnesium nutrition as affected by salinity (Grattan & Grieve, 1994). Calcium is a strong competitor of Mg, and the binding sites on the root plasma membrane appear to have less affinity for Mg than for Ca (Marschner, 1995). Thus, high concentrations of Ca often result in increased leaf-Ca along with a marked reduction in leaf-Mg (Bernstein & Hayward, 1958). For example Ruiz *et al*. (1997) found that NaCl salinity reduced leaf Mg concentrations in citrus. However increases in salinity are not always associated with decreases in leaf Mg. Bernstein *et al*. (1974) found that increases in salinity ($NaCl + CaCl_2$) only reduced leaf Mg concentration in beet and had little or no effect in leaves from five other vegetable crops that they examined.

It has been known for several decades that solutions with a Mg/Ca ratio greater than one, such as those that result by diluting sea-water, reduces the growth of maize (Key *et al*., 1962). In eucalyptus, Mg-salts were found to reduce root growth more than Na-salts (Marcar & Termaat, 1990) and this effect was associated with low concentrations of calcium in the root. Calcium-induced Mg deficiency has been observed in sesame (Nassery *et al*., 1979).

2.5 Runoff and drainage from mine water irrigated fields

Runoff and drainage could be the main means of salt transport from coal-mine water irrigated fields to water resources. A rainfall event that is greater than the water holding capacity and is greater than the infiltration rate of the soil initiates surface runoff, which carries salts watercourses. Drainage that occurs through natural lateral flow or vertical percolation of excess water below the root zone could also be another means of salt transport. The salinization of water resources through drainage and runoff, therefore, could be a major concern regarding the sustainability of irrigation with coal-mine water.

Factors influencing runoff and drainage

Runoff

Several factors can affect surface runoff, such as precipitation (amount, intensity and duration), soil type, soil water content, vegetation and topography (Mishra & Singh, 2003). Infrequent torrential rainfall easily erodes salts from the soil surface, while soft drizzly rain infiltrates into the soil resulting in minimal salt transport by surface runoff. Porous soils such as sands are well-drained soils which can absorb water more quickly than fine-textured (clay) soils and have a lower runoff potential than poorly-drained soils (less-porous). Antecedent soil water content also is very important in runoff generation, as wet soils generate more runoff than dry soils (Gómez-Plaza *et al*., 2001). Topography is an additional factor affecting water velocity, infiltration rate, and overland flow rate. Cropping promotes slope stability, and reduces adding salt and sediment load into streams. Runoff can be minimized by increasing soil surface storage and by increasing the infiltration rate of the soil, by leaving crops residues as well as mulching.

Drainage

Drainage occurs when the plant/soil system is unable to use or store the amount of water it receives over a period of time. Rainfall, soil properties and vegetation affect the extent of drainage.

Soil properties such as clay mineralogy, clay content (or texture), CEC/clay ratio, bulk density, soil structure, porosity, hydraulic conductivity and water holding capacity are key determinants of drainage (Silburn & Freebairn 1992; Keating *et al*. 2001, 2002; Yee Yet & Silburn 2002, 2003). For instance, drainage tended to be highest at low clay contents, lowest at medium clay contents and intermediate at high clay contents. Sandy soils drainage is usually higher than for clay soils.

Cropping system also affects the pattern of soil water use and storage (Freebairn *et al*., 1986, 1996). Deep drainage is generally greater under annual crops and pastures than native perennial vegetation (Walker *et al*., 1999; Cocks, 2001; Heng *et al*., 2001). Management of soil surfaces (tillage) and crop residues (stubble) also affects drainage. Evidence of greater solute movement under zero tillage than under conventional tillage has been noted in a number of studies (Dalal 1989; Turpin *et al*., 1998; Turpin *et al*., 1999; McGarry *et al*., 2000). Modelling studies (Walker *et al*., 2002 ; Keating *et al*., 2002) have compared farming systems in terms of their susceptibility to drainage. They generally find drainage under annual wheat > (greater than) annual sorghum> perennial pasture>native vegetation in Australia.

Reduction of drainage in rainy seasons could be difficult as it is dependent on the rapid development of annual crop root systems. However, perennial species such as trees generally have deeper rooting systems which can be much more effective in abstracting soil water and reducing drainage (Huda & Ong, 1989). Since trees have deeper root systems than annual crops and use water outside the rooting zone of annual crops, they have been used as companion species for crops in agroforestry systems.

Runoff and drainage measurements

Runoff and drainage quantity and quality measurements are necessary to quantify the magnitude of the salt loads from coal-mine water irrigated fields. Runoff quantity and quality can be measured by erecting runoff weirs at the lowest end of the irrigated field, where the runoff water converges. Since the carrying out of field experiments to measure salt transport and design appropriate management solutions is expensive, different techniques are used to estimate runoff quantity and quality. The most commonly used is the Soil Conservation Services Curve Number (SCS-CN) method which was developed in 1950 by the United States

Soil Conservation Services (US-SCS) (Mishra & Singh, 2003). This method is characterized by the following equation:

$$Q = (P - I_a)^2 / (P - I_a + S)$$

Where

Q is Runoff (mm)

P is Precipitation (mm)

I_a is Initial Abstraction (stored, intercepted, and infiltrated water) (mm) and approximated as 0.2S, S is a parameter derived from the following equation where

$S = (1000/CN) – 10$

CN is Curve number

The equation simplifies to:

$Q = (P - 0.2S)^2 / (P + 0.8S)$

CN is the slope of the line between rainfall and surface runoff. The US-SCS determines the values for these curve numbers. They are derived from hydrologic soil group, land use and antecedent soil water content conditions.

Soils are divided into four hydrologic soil groups. Group A has low runoff potential (i.e., runoff is unlikely), having a final infiltration rate of > 7.62 mm hr^{-1}. Group B has moderate infiltration rates when wet, having final infiltration rates between 3.81 and 7.62 mm hr^{-1}. Group C has low infiltration rates when wet (i.e., is likely to provide surface runoff), having infiltration rates between 1.27 and 3.81 mm hr^{-1}. Group D has a high runoff potential, having infiltration rates < 1.27 mm hr^{-1} (SCS, 1971). Antecedent soil water content conditions assess how wet the soils were before the storm. The higher the antecedent soil water content, the greater the surface runoff. This SCS-CN approach, however, does not consider the quality of runoff.

Drainage can be measured using direct methods, for instance, lysimeter, which is a device to measure the volume of the percolating past the bottom of profile flow of water with or without application of tension, or to obtain water samples from the soil (Titus & Mahendrappa, 1996).

Indirect methods include using Darcy's law (Bond, 1998), salt balance, water balance (Zhang *et al*., 2002; Ward *et al*., 2001), groundwater response (Cook & Herczeg, 1998; Allison & Hughes, 1983), the hydraulic water potential gradient and soil hydraulic conductivity (Jury *et al*., 1991), soil water balance modelling (Annandale *et al*., 2006; Rhoades & Loveday, 1990; Zhang *et al*., 2002) and chloride balance (Lidón *et al*., 1999). Annandale *et al*. (2006), used boreholes drilled inside and in close proximity to the mine irrigated fields, to measure salts moving through a profile. Accurate determination of drainage using a water balance (Wagenet, 1986) relies heavily on how accurately the evapotranspiration can be measured or estimated. Evaporation when not limited by water deficits or other crop growth limitations, runoff and drainage can be estimated with reasonable accuracy using climate data and crop coefficients (Doorenbos & Pruitt, 1984; Jensen *et al*., 1990; Allen *et al*., 1994).

Annandale *et al*. (2006), for instance, used a mechanistic soil water balance model to estimate leachate from coal-mine water irrigated fields to investigate the impact of large scale irrigation on groundwater resources.

The hydrological cycle plays a dominant role in the movement of salts. Drainage and runoff measurements/estimates are, therefore, very site and season specific, varying from year to year depending on the total amount of rainfall, but also on its seasonal distribution. Extrapolation of field measurements is further complicated by the diversity of soils and crops, and the lack of information on the interaction between crop, soil and climate variability as they affect water use and water loss.

Beven, 1989 and Wagener *et al*. 2001, have reviewed a large body work on runoff hydrology. Their study suggests that physically based models cannot predict runoff generation in the field adequately as they are not good descriptors of runoff processes, except under some special circumstances. In the assessment of irrigation with coal mine water for large scale irrigation, a reliable runoff model is required to ascertain whether surface waters are impacted. Crop models are believed to be effective tools in the extrapolation of research findings over time, soil type and climatic region. However, the acceptance of outcomes from simulation studies is dependent on the confidence in the models used to predict crop growth, water use, soil water dynamics and deep drainage. Soil water balance model output coupling with groundwater and surface water models, could be helpful in the assessment of water resource pollution induced from agriculture. Therefore, modifying the runoff component of the SWB model is an appropriate method for this study given the available data, goals of the study and goals of the larger research program.

Possible impact of mine water irrigation on surface water

This section focuses on the aspects that were directly related to the objectives of the research. Therefore, it includes a review on the impact of mine water on the water resources of the Olifants Catchment.

There are a large number of mining operations exploiting a wide variety of minerals in the Olifants Catchment. Available evidence suggest that lime treated AMD and AMD leakages are likely to be a threat to water resources, especially to the water quality of all streams and rivers (Vermeulen *et al*., 2008). The largest impact of freely releasing lime treated AMD onto the environment could be to salinization of the water resources. Whereas AMD that leaks from closed or abandoned mines have a serious impact on the productivity of ecosystems by affecting biological organisms within the streams (IIED, 2002). One of the worst features of AMD could also be its persistence in the environment and it has the potential for severe long-term, (possibly several decades long (IIED, 2002)), impacts on surface and groundwater, and on aquatic life.

This serious impact caused by mining or attributable to mining has been the subject of concerted research and management for several decades in South Africa. Coaltech 2020 is a collaborative research programme which has been formed by the major coal companies, Universities, CSIR, NUM and the state to address the specific needs of the Coal Mining Industry in South Africa using local and international knowledge and skills. This is one of the programmes that is attempting to derive appropriate and cost effective management strategies that will help resolve these problems.

As part of this programme, Annandale et al. (2006) and Vermeulen *et al*. (2008) investigated the impact of irrigation with mine water on groundwater resources for the first time at field scale in southern Africa. Out put of the SWB model was used during the groundwater modelling. According to Annandale et al. (2006), irrigating large areas with gypsum rich mine water could be feasible and sustainable if careful attention is paid to the specificity of each situation. They also advised that large errors can be made in designing such irrigation schemes if the amount of deep drainage leaving the root zone, the storage capacity between the base of the root zone and the underlying aquifer systems, and the hydraulic characteristics of the aquifers are not properly matched. Percolation from irrigation in excess of what the underlying aquifers can transmit from the site, will lead to rising water tables, and over time, water logging and salinization of the root zone. This will necessitate the installation of expensive drainage systems, or ultimately, result in the failure of the irrigation scheme.

Vermeulen *et al*. (2008) also reported that the overall water quality trend in the deeper aquifer indicated no significant water quality deterioration over the monitoring period. Some exceptions occurred on a very sandy soil, with consistent water quality degradation, but none of the boreholes outside the pivot areas show any meaningful change in water quality due to leaching from irrigated area. In the short to medium term, the evidence from groundwater monitoring shows that irrigation with mine water does not hold significant threats to the regional groundwater quality. The hydraulic and attenuation factors preventing the salts in the mine water used for irrigation from being mobilized down the soil profile and into the aquifer are important considerations in this process. From this study they concluded that irrigation with gypsiferous mine water, if properly managed, could seriously be considered as part of the solution towards the challenge of responsible management of the considerable volumes of mine water available during mining and post closure

Saline water irrigated fields could generate runoff salts during large rainfall events. The magnitude of runoff salt depends on the soil type, slope and rainfall intensity and soil salinity (Gilfedder & Walker, 2001; Rhoades *et al*., 1997). Thus, the salt discharge by surface runoff from mine water irrigated fields needs to be quantified and used to validate models like SWB to better understand the impact of large-scale irrigation on surface water resources.

Knowledge gap

In conclusion, a large body of knowledge exists regarding the irrigation of crops with saline and sodic waters. The use of saline water for irrigation requires selection of salt tolerant crops, sound irrigation water management and the maintenance of favourable soil chemical and physical properties to ensure adequate infiltration and salt leaching. However, there will be several uncertainties when it comes to crop and soil response, to the long-term impact of irrigation with the unusual water qualities emanating from coal-mines.

Several of the studies available in the literature have been done using of saline irrigation waters that mainly consist of NaCl as the salinization agent (Grattan & Grieve, 1999). A number of studies have also examined crop response under solutions of various anionic compositions, particularly SO_4 and HCO_3, in controlled conditions in glasshouses. Coal-mine water is usually saline water that can be of various compositions of $CaSO_4$, Na_2SO_4, $MgSO_4$ or $NaHCO_3$ and commonly is dominated by cations such as Ca, Mg and Na, and anions such as SO_4, HCO_3 and Cl. In view of these uncertainties the literature could not answer all the research questions as these waters are atypical of waters used in most studies. Thus, an assessment of the suitability of poor quality mine waters for irrigation and its long-term impacts on crops and soils is worth investigating in view of possible future uses of these mine waters.

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