Soils and the environment: the past 25 years

J. L. Schoeman & P. W. van Deventer


To link to this article: https://doi.org/10.1080/02571862.2004.10635070

Published online: 15 Jan 2013.

Submit your article to this journal

Article views: 610

View related articles

Citing articles: 5 View citing articles
Soils and the environment: the past 25 years

J.L. Schoeman¹ and P.W. van Deventer²

¹ARC-Institute for Soil, Climate and Water, Private Bag X79, Pretoria 0001, Republic of South Africa
²Envirogreen, P.O. Box 20813, Noordbrug 2522, Potchefstroom, Republic of South Africa

Local work on selected environmental impacts on soil quality and the role of soils in the attenuation of environmental pollution is reviewed. Examples of negative impacts from mining, industries, urbanization, agriculture and forestry abound. The safe and sustainable use of various waste products may impact positively, but norms and standards are needed. Soil nutrient deficiencies or excesses give rise to nutrition-related disorders in humans and animals. Through bioremediation, soil becomes an ally in restoring environmental health. Harmonization of agricultural production and the environment is a political target towards which science must provide effective decision support. The development and application of a national environmental monitoring and evaluation system is needed for incorporating environmental data, information, norms and standards into a holistic picture. There is a need for improved cross-linking and synergy between various sectors impacting on the environment. The local government level is becoming critical to environmental management. The safe and sustainable use of sewage sludge/fly ash combinations on agricultural soils is suggested to be a highly promising future avenue of environmental R&D.

Keywords: Soil degradation, environmental degradation, soil pollution, pollution attenuation

Introduction

The information era brought a clearer focus on the finiteness of global resources, the fragility of our ecosystems and the need for their sustainability. Soil science as a discipline oriented itself rapidly to an environmental focus and started contributing to it. Not only did a new branch of soil science develop - environmental soil science - but all branches of soil science underwent a measure of reorientation towards the sustainability of the environment. Conservation tillage is an example.

The extensive interface between soils and the environment made it difficult to delineate the scope of this paper, the theme of which is the impact of human activities on soil resources and the role of soils in the attenuation of environmentally harmful substances. Impacts resulting from ordinary agricultural land use practices (e.g. soil erosion, nutrient depletion, the decline in soil structure) are dealt with in various other papers in this issue. There will be meaningful soil environmental work not being addressed here due to confidentiality preventing it from being published. There will also be meaningful work that has been omitted through oversight. For the latter, sincere apologies are offered.

Modern day environmentalism started in the United States in the mid-1960s with the onset of large-scale environmental awareness by the general public. The mid-1970s may perhaps be regarded as the beginning of the environmental era in South Africa. Subsequently, environmental issues gained so strongly in importance that Section 24 of the Bill of Rights in the Constitution of South Africa (Republic of South Africa, 1996) claims that “everyone has the right: (a) to an environment that is not harmful to their health or well-being; (b) to have the environment protected, for the benefit of present and future generations, through reasonable legislative and other measures that (i) prevent pollution and ecological degradation; (ii) promote conservation; and (iii) secure ecologically sustainable development and use of natural resources while promoting justifiable economic and social development.” Few people would argue against soil health being basic to a healthy environment.

Blum (2002), however, warned that the role of soils in sustaining society and the environment is complex and not well understood by the broad public, in politics and in decision making. In addition to its long recognized role of producing biomass for agriculture and forestry, the soil filters, buffers and transforms harmful substances, protecting the food chain and water against pollution. Soil is the most important gene reserve, containing more biota in species diversity and quantity than all above-ground biomass.

At congresses of the Soil Science Society of South Africa (combined congresses with other societies excluded), the percentage of papers dealing with the subject matter of this paper increased steadily and linearly from one in 1980 to twenty in 1999.

Environmental impacts on soils due to agriculture

The environmental impacts of agriculture on soils are numerous. This paper only touches on three, namely the impact of irrigated agriculture where polluted water is used, soil pollution as a result of agricultural practices, and the impact of agriculture on soil biology.

Soil degradation as a result of irrigation with water of deteriorating quality

Despite its obvious advantages, irrigated agriculture continues to give rise to soil degradation. This takes the form of salinization, sodicity, waterlogging, structural breakdown, crustning and compaction. Contributing causes are deteriorating water quality, low suitability of the soils, sub-optimal management, poor planning and the indirect consequences of economic pressures. Mainly those impacts arising from an accelerated decline in water quality due to mining and industries are addressed.

Salinization

Streutker (1989) estimated the area of South African land permanently salinized and out of production at about 40 000 ha and the area affected by temporary salinization at another 60 000 ha. It is probably correct to assume that in the past, mainly soils and landscapes of poor suitability were affected.
Salinity profiles in the soil were nearly uniform and generally recovered fairly rapidly after installation of drainage. Currently, the main threat results from increasing use and re-use of water resources (Du Plessis, 1991; Johnston, 1991). Salinization of water, and consequently, salinization of the better quality soil resources used for irrigation thus poses the future threat.

A study of the salinity, sodicity and waterlogging status of different soil types irrigated along the Vaal River with its marked decline in water quality downstream (Le Roux, 2001; Du Preez et al., 2000), illustrated some effects that can be expected to occur more widely in future under conditions of deteriorating water quality. The following trends were found: the electrical conductivity of the saturated soil extract (ECe) of well drained sands tended to equilibrate between 50 and 100 mS m⁻¹ (indicating low salinity); sands above water tables became saline; the build-up of salinity and sodicity in irrigated clays reflected differences in the soil's ability to undergo leaching. The warning is sounded that limited leaching of irrigated soils poses the threat of the root zone becoming saline and sodic. In this regard, Du Plessis (1991) pointed out that insights into the way crops respond to the type of soil salinity arising from deteriorating water quality have improved during the past 25 years and that it has become clear that crops may be irrigated with water of higher salinity, allowing the salinity at the base of the root zone to rise to higher levels than previously thought possible.

De Clercq et al. (2001), experimenting with full scale and supplementary irrigation of vineyards with saline water of which electrical conductivity (EC) values varied between 75 and 500 mS m⁻¹, using micro jet and various frequencies of drip, found the following: soil profile salinity varied with cultivar, time of the year, soil depth and type of irrigation; average EC, values during the trial period of 5 years varied between 50 and 225 mS m⁻¹; and subsurface drip treatments produced greater accumulations of salts at the soil surface than surface water applications.

Gypsiferous mine water

Large volumes of acid and neutralized waters have recently become available from the mining industry (Du Plessis, 1983; Scotney & Van der Merwe, 1991; Tanner et al., 1999). This could contribute additional water for irrigated agriculture once research is undertaken and policy is reviewed (Scotney & Van der Merwe, 1991). After treatment with hydrated lime and precipitation of the major portion of the CaSO₄ so formed, the water is saline (gypsiferous) with EC values in the order of 130 to 290 mS m⁻¹. Du Plessis (1983) postulated on the basis of chemical equilibrium modelling that lime-treated acid mine water may be suitable for irrigation. The limited solubility of gypsum would cause its precipitation in the soil, thereby limiting the effective soil salt concentration to which the plant is subjected. No serious sodium-related soil physical problems were predicted due to the relative increase in sodium caused by calcium precipitating. Jovanovic et al. (1995; 1998; 1999; 2000) used treated acid mine drainage water in irrigated field trials. They found that a wide range of moderately sensitive crops may be successfully grown using water with EC values of 132-200 mS m⁻¹ (the main ionic species was Mg²⁺). Resultant increases in soil salinity were reported as being acceptable, the ECe and the pH of the deeper rooting zone remaining low after two years. Fifty-year simulations of the salt balance indicated that between 23 and 100% of the added salts would be leached. It was concluded that salt leaching and contamination of groundwater may be minimized by adopting a deficit irrigation strategy leaving room for rain to refill the soil profile. A simulation study in Botswana with gypsiferous mine water with an EC of about 310 mS m⁻¹ (Jovanovic et al., 2001) led to the conclusion that under the particular climatic and soil conditions of Selibe Pikwe, large amounts of effluent mine water can be successfully disposed of through irrigation. Between 18 and 32% of the total amount of salts added through irrigation was predicted to leach after 11 years, the remainder being precipitated in the soil profile in the form of gypsum. A slow process of gypsum dissolution and leaching by rainfall was predicted after the cessation of irrigation with mine water. This means that large quantities of salt can be immobilized in the soil profile, removed temporarily from the water system, and released in small amounts into the groundwater over an extremely long time period. Grobler et al. (1999) predicted gypsum precipitation in mine soils by means of simulated time scenarios of irrigating soil columns with gypsiferous water. Nepumbada et al. (1999), examining the influence of gypsamiferous water on soil aggregate and colloid stability, found an initially rapid, followed by a steady, increase in soil aggregate stability on gypsum treatment. A rehabilitated mine soil, however, showed a slight decrease in aggregate stability.

Sodicity, structural degradation and surface sealing

Sealing of the soil surface under rainfall or irrigation takes place as a result of the kinetic energy expended by water drops, taking into consideration the composition of the water and the chemical, textural and mineralogical composition of the soil.

Van Deventer (2000) investigated the effect of water of different EC and sodium adsorption ratio (SAR) values on surface sealing and infiltration, using soils differing in clay mineralogy but with similar textures. The kinetic energy of the water drops was kept constant. When used on kaolinitic soils, water with low EC gave rise to deflocculation, causing surface sealing and a reduction in infiltration. These soils remained stable when irrigated with water with higher ECs, even at high SAR values. Illitic soils reacted similarly to kaolinitic soils to the effects of EC and sodium up to an exchangeable sodium percentage (ESP) threshold of approximately 10, above which the infiltration rate decreased rapidly. Smectite-dominated soils were found to be particularly sensitive to the SAR of the irrigation water and underwent severe deflocculation when EC values were low and the SAR high.

Surface sealing due to water drop impact from centre pivot or sprinkler irrigation systems is recognized as a severe problem. This is due to negative effects on the efficiency of the systems and environmental pollution through agricultural chemicals dissolved in runoff water. The importance of reducing the energy flux density of the water by management of the application rate, median drop diameter and fall height is being increasingly recognized (Liengme, 1994). Scanning electron microscope studies (Liengme, 1995) showed that crusts were well established (at 0.8-1.2 mm depth) after four
high-energy irrigation applications. Subsequent irrigation applications, though appearing to have little effect on the further development of the crust, were strengthening rather than destroying it.

Waterlogging

Du Plessis (1991) expressed the view that although waterlogging appears to be an incessant problem countrywide, and especially serious on some irrigation projects (e.g. Golden Valley and Vaalharts before the installation of drainage), the current situation seems to be largely under control from a national perspective.

Monitoring

Due to elevated soil salinity levels expected in future, it will become increasingly necessary to monitor the situation on irrigation schemes in order to timely identify salinization trends and potential problems. In this regard, advances were made in the use of the four-electrode probe and electromagnetic induction soil conductivity sensors.

The four-electrode probe has been shown to offer a cost and time effective means of measuring the salinity level in soils, provided that the necessary calibrations are done (Johnston, 1991). As it is physically difficult to insert the probes deeply into the soil, Johnston et al. (1995) tested electrode configurations which included inserting electrodes in an array into the surface soil only and found good agreement between the methods tested in the upper 0.9 m of the soil, but that the surface array method over-estimated EC$_c$ (conductivity of the bulk soil) by 35% at 0.9 to 1.2 m depth.

Johnston et al. (1994) investigated the use of the EM-38 electromagnetic induction soil conductivity sensor (specially developed for soil salinity work) and concluded that the evaluation of the soil salinity/sodicity status of irrigated areas can be greatly enhanced by employing electromagnetic induction measurements (particularly with the aid of GIS presentation) to rapidly delineate categories of problematic soils. The information, in conjunction with soil type, may be used as a rational basis for selecting a limited number of suitable sites for soil sampling (Johnston et al., 2001). By using the instrument in either a horizontal or vertical position, the depth at which salinity occurs can be assessed. When used in the horizontal position, it is most responsive to upper layers with 70% of the response from the upper 0.75 m of the soil; when used in the vertical position, it is more responsive to deeper layers, with 68% of the response from the upper 1.5 m.

Paterson et al. (1998) tested the EM-38 sensor in conjunction with ground penetrating radar for investigating saline soils with water tables. Due to variation in soil types and the types of salt present, the EM-38 has to be carefully calibrated by means of supportive sampling. Regression equations have to be prepared for each study area. It was concluded that the method is suited to rapid reconnaissance sampling as opposed to detailed studies. It is suited to monitoring actions and can be automated, in conjunction with GPS apparatus to improve positional accuracy.

At a workshop on a South African Strategic Irrigation Information and Monitoring System (ISCW, 2000), it was reiterated that a national irrigation monitoring system is needed to provide up-to-date information on the state of land and water use for irrigated agriculture. Such information is required for policy development and to meet the requirements of international treaties and conventions aimed at ensuring sustainable development. Satellite remote-sensing techniques, having been developed into effective methods of land-related data acquisition, are expected to be a component of the envisaged system. The National Land Cover database (ISCW, 2000; Nell & Van den Berg, 2001) contains the spatial distribution of irrigated land and is a small first step in the direction of a monitoring system.

In the absence of a formal monitoring system, the impacts that water of suspect quality might have on land is determined and monitored in an ad hoc way. This is mainly in order to supply data for obtaining permits for the release of mining or industrial water into public streams. An example is the six-year old monitoring programme of irrigated agriculture along the Blesbokspruit on the East Rand (Nell, 2000; 2002). Through measuring soil salinity and other soil chemical properties at regular intervals, fluctuating trends in soil salinity and sodicity were observed. EC$_c$ values at the bottom of the root zone increased from 70 mS m$^{-1}$ in 1996 when mine water was first released and monitoring started, to 177 in 1999 (dry year) and dropped again to 95 mS m$^{-1}$ in 2001 (wet year). Long-term trends were masked by seasonal events such as high or low rainfall, high or low irrigation demand and the extent of winter irrigation. No noticeable adverse effects have been noticed on crops. The range of crops that could be grown did not change.

Soil pollution as a result of agricultural practices

The use of agricultural pesticides, fertilizers and organic soil ameliorants

Hazardous wastes from agricultural operations consist mostly of pesticide and fertilizer runoff and leachate.

The Olifants River in the Mpumalanga and Limpopo provinces is generally regarded as one of the most polluted rivers in Southern Africa. The main pollutants deriving from agriculture are pesticides, phosphorus and nitrogen. An estimated 24% of the catchment above the Witbank Dam is used for maize under monoculture (Van Niekerk, 1992). Pesticide pollution impacts seriously on wildlife, as is shown by the fact that fish eagles resident in the Loskop Dam area have been found to have the highest pesticide levels accumulated in their eggs when compared to data from around South Africa (Batchelor, 1992). The threat to soils, however, is limited.

The concentration of total phosphorus in water of the Olifants River was reported by Van Niekerk (1992) not to have changed over the decade preceding 1992. Based on monitoring conducted during 1990/91, it was stated that of the 75 tons of P that derives from the Witbank Dam catchment, 40 were derived from natural weathering and agriculture. Of the 118.3 tons of N derived from the Witbank Dam catchment, only 22.3 tons were estimated to originate from agriculture (Van Niekerk, 1992). Under most circumstances, phosphorus and nitrogen pollutants in irrigation water would not be considered as posing a serious threat to irrigated soils although excessive nitrates may result in too much vegetative growth and delayed ripening of crops.
Soils may be polluted by heavy metals contained in fertilizer. Organic soil ameliorants may decrease or enhance the bio-availability of heavy metals present in the soil. Van der Watt & Van der Walt (1994) found that the presence of coal-derived fulvic acid led to increased mobility of Ni, Fe and Mn and decreased mobility of Cu. McLaughlin & Hamon (2001) pointed out that from a food chain perspective, perhaps the most important inputs of Cd to soil are those that occur as a result of fertilization with either inorganic or organic fertilizers. P fertilizers are the main source of fertilizer-derived Cd. Cd is largely present in insoluble forms in the rock, but is converted into water-soluble forms during fertilizer manufacture. Trace element fertilizers and organic fertilizers may contain Cd as an impurity. Where wastes such as sewage sludges are used, additional inputs of Cd to agricultural soils result.

Truter & Rethman (2001) investigated the use of a combination of sewage sludge, fly ash and lime (SLASH) as a fertilizing soil ameliorant in pot and field trials. The rationale was that enormous quantities of coal-fired power station fly ash and sludge are produced in close proximity to acidifying cropland which, separately, have limitations which can partly be overcome if the products are used together with lime at soil pH levels that are high enough for toxic heavy metals to be immobilized. At 5% SLASH application (100 t ha⁻¹), yields of various crops and pasture species were approximately 440% better than in the control. The pH increased by 1.5 units. The two potential problem heavy metals, Ni and Cd, as well as the micronutrient B, were within safety specifications based on leaf analysis.

Elevated nitrate levels in ground water may result from agricultural activities (ploughing of soils resulting in a breakdown of accumulated organic matter and consequent release of nitrate, feedlots, irrigation, dryland fertilization and manure and sludge application). Concern caused by nitrate in water centres largely around human and stock health (Tredoux & Du Plessis, 1993). Nitrate-bearing water has limited use. When used for irrigation, it can result in excessive vegetative growth. Nitrate-enriched groundwater entering open water bodies may undergo denitrification by microorganisms present in organic-rich bottom sediments (Harck et al., 1997).

**Feedlots**

Application of waste and effluent from feedlots and piggeries to agricultural land is mostly beneficial, as soil physical and biological properties as well as fertility are improved. However, it may also result in salinization and elevated levels of Cu, Zn and Hg in soil and water if applied excessively.

Steyn (1994a), in evaluating a number of case studies of piggeries and cattle feedlots, concluded that the biggest potential source of pollution is sludge ponds. These may be highly saline with elevated levels of potentially toxic elements including Al, Co, Cr, Ni, Se and Mo.

**Agriculture and soil biological diversity**

Concerns are mounting about the effects of monocultural agronomic practices on the soil biology in South Africa and its diversity.

Dlamini et al. (2001) reported on the effects of land use on the size and composition of the earthworm community in northern KwaZulu-Natal. It was found that earthworm numbers followed the order: burnt sugarcane < trashed sugarcane = grassland = eucalyptus plantations = pine plantations = wattle plantations = avocado orchard < citrus orchard < banana plantation < native forest < kikuyu pasture. The pattern of change in organic C, and particularly microbial biomass C, with land use showed broadly similar trends to those for earthworm numbers. It was demonstrated that the C input to the soil and the amount of labile metabolizable C present are the major determining factors to the size of both the soil microbial and earthworm communities. It was also found that native earthworm species accounted for less than 20% of the total number of earthworms collected, but that these dominated under grass veld. A highly adaptable South American species, dominating cultivated fields in many tropical and subtropical parts of the world, was present in significant numbers under all land uses and made up to 50% of the communities under sugarcane, pine and wattle. Other exotic species originated from India. It was concluded that the size and composition of earthworm communities are greatly affected by changes in soil and land management.

Mile et al. (2001) studied the condition of sandy, degraded soils under pasture in the Tsitsikama, employing chemical and biological indicators. The damage of re-cultivation and re-sowing of pastures with limited longevity to soil biota was pointed out. It was re-emphasized that long-term kikuyu pastures increase the soil organic matter content in comparison to veld, with accompanying improvements in soil biological activity, and in some cases, physical conditions. Already degraded soils can be remedied effectively by converting them to permanent kikuyu pasture.

**Soils and carbon sequestration**

The exchange of carbon between the atmosphere and biosphere through photosynthesis and respiration is increasingly being recognized as important in influencing global warming and climate change. Anthropogenic processes such as arable land use and land use change are known to alter the naturally occurring exchange processes and rates.

Du Toit et al. (1994) found that rainfed cultivation results in a 10 to 75% decrease in soil organic matter and that the rate of loss is high during the first five years of cultivation, decreasing until equilibrium is reached after ca. 35 years of cultivation, after which little or no further loss occurs. Du Preez & Wiltshire (1997) reported increases in soil organic C in some irrigated soils over time. Barnard & Newby (2001) concluded that cultivation in South Africa generally resulted in halving the carbon present under grassland and recommended that a protocol be developed for the selection of benchmark sites as well as sampling and analytical procedures for soil organic matter monitoring.

**Agricultural threat to peatlands**

Peatlands, as a soil resource, are mainly impacted on by mushroom farming. Mushroom growers in South Africa use peat as substrate since no other cost-effective alternative exists at present. More than 90% of the peat utilized in the mushroom industry is mined locally (Marneweck et al., 2001).
Environmental impacts on soils due to forestry and deforestation

Nutrient cycling, leaching and acidification

James & Hughes (1998) investigated the effects of pine plantations on soils of the coastal dune system at lake St. Lucia which were previously under secondary grassland, and found that the pine plantations allowed the indigenous dune forest to re-establish itself as an understory. The latter ameliorated the acidifying effects of the pine plantation on the soil, resulting in higher pH and higher soil fertility than where the plantation is without it. Tarassova et al. (1999) compared the Cu, Zn and Fe content of highly weathered, potentially trace element deficient soils under yellowwood, redwood and pine plantations in the Magoeaskloof area and found bio-accumulations of these elements in the top 200 mm of the soil, associated with organic matter. Total analyses showed no correlation with available forms. Differences due to tree species were not apparent. Maiterechera (1999) proposed the re-introduction of *A. erioloba* in an agroforestry land use system for North West Province, based on various beneficial roles it can play. Positive effects on the soil would be a reduction in wind erosion and the fixing of nitrogen.

Du Toit & Fey (1992) compared soil acidity and lime requirement analyses (topsoils) from 40 plantation sites (covered by 10 to 60 year old *Pinus* spp., *Eucalyptus* spp. and *A. mearnsii*) and adjacent grassland sites. The pH values of afforested soils were found to be significantly lower than those of grassland soils. Between 3.5 and 9.6 t CaCO₃ per ha would be required to restore the plantation soils to the condition under grassland. This translates to an annual rate averaging approximately 150 kg CaCO₃ per ha. Bases are also exported in harvested products (Fey et al., 1990).

Nowicki & Fey (1997) investigated the impact of the acidifying effect of tree plantations on the composition of soil solutions and surface waters. A key issue is the influence on the mobility of metals, particularly Al and Mn. Soil solution Al under forest, at 0.5-2 mg L⁻¹ was found to be enhanced 2 to 40-fold relative to that in the adjacent virgin soil. In afforested catchments underlain by poorly buffered quartzitic rocks, elevated Al (up to 0.3 mg L⁻¹) was recorded in acidified stream waters. In the majority of soil solutions, Mn did not exceed 5 mg L⁻¹.

Water repellency

Water repellency in soils may be attributed to coatings on the soil particles of hydrophobic substances or organic origin. It may be related to fungal activity in the soil, to vegetation type or vegetation age. Fire may exacerbate water repellency in the soil. It is more pronounced when the soil is dry.

Scott (1989; 1992) reported on investigations into the phenomenon of water repellency in soils under plantation and indigenous forest cover. Representative soils from several important timber-growing areas in South Africa were sampled from beneath different timber plantation types and adjacent natural vegetation, and subjected to a range of standard wettability tests. It was found that *Eucalyptus*, wattle and indigenous forest in general, relative to other vegetation types, induce a high level of water repellency in the soil. The potential effect of water repellency may not be noticeable when the soil is wet or covered with plant litter offering surface storage of water. When the canopy and ground cover are removed during clearfelling or as a result of fire and the soil is exposed to drying, the site is at risk of overland flow occurring during rainstorms, leading to soil erosion and reduced soil water replenishment.

Environmental impacts on soils due to mining

Mining activities and wastes are a potential source of soil degradation. The main factors responsible for environmental degradation are: oxidation of iron pyrites contained in mined materials, the presence of toxic substances and other pollutants in minerals and groundwater, high concentrations of dissolved and suspended solids in mine effluents, toxic reagents used in metallurgical processes and erosion from residue dumps and slimes dams (Korentajer, 1992). To these may be added compacted and/or polluted footprints of reworked dumps and slimes dams, and damage to soil resources due to strip mining.

Van Deventer & Le Roux (2001) identified a need for expanding the South African soil classification system to better describe the anthrosols resulting *inter alia* from mining. Five new diagnostic horizons were proposed.

Acid mine drainage

Acid mine drainage (AMD) results when water percolates slowly through old mine workings, coming into contact with exposed and oxidized pyrites (Thompson, 1980). Mine drainage water may have very low pH values, may contain substantial amounts of heavy metals, in particular Fe, Mn and Pb. Toxic concentrations of other pollutants, particularly F, may occur. Salinity levels (particularly sulphates) may be high.

Mining activities (in conjunction with coal-fired electricity generation) in the Olifants River catchment result in an increasing sulphate concentration in the Witbank Dam. According to Van Niekerk (1992), the average annual sulphate mass exported from the upper-Olifants river to the Witbank Dam was estimated to be 12 300 t SO₄⁻². The following were main contributors (t SO₄⁻²): natural and anthropogenic weathering, atmospheric deposition and agriculture (2 440 or 19.8%); municipal sewage treatment plants (387 or 3.1%); power station effluents (796 or 6.5%); coal mining point sources (320 or 2.6%); and coal mining diffuse sources (8 373 or 68.0%).

The total dissolved solids (TDS) concentration of Witbank dam water increased from 169 mg L⁻¹ in 1978 to 270 mg L⁻¹ in 1991 (Van Niekerk, 1992). The headwaters of the Olifants River were described as a Ca/Mg bicarbonate water and at the entry into the dam as a CaSO₄ water. The typical winter base flow SO₄₂⁻ concentration is 300 mg L⁻¹ and that during the wet summer months, 100 mg L⁻¹. The bulk of the pollution mass, however, enters the Witbank Dam during the wet summer months (Van Niekerk, 1992). The effect of this salt load on the soils and crops of Loskop Irrigation Scheme is unknown (A. Venter, personal communication).

In a study on the presence of metals in the water, sediment and fish of the middle and lower Olifants River, Grobler et al. (1994) found low levels of metals, both in the soluble and acid extractable form, in water and fish. It was postulated that the low levels in dissolved form and in fish may be attributed
to suspended sediment providing binding sites for metals, making them unavailable to aquatic organisms. The metal contents of sediments varied over the catchment in accordance with mining activities. It was concluded that generally, the levels found represented the natural geological background but that elevated levels of some metals do exist at localized points. This was corroborated by Van Niekerk (1992) stating that heavy metal concentrations are currently not of concern in the Witbank Dam.

Häßlich et al. (1997) reported on the biogeochemistry of AMD from coal mining operations in the Witbank area. Diseased underground collieries decant AMD (pH 2.6-2.9, TDS 1750-3070 mg L-1) in large volumes into the Blesbokspruit catchment. Sediment, water, secondary mineral precipitate and algae samples were used to characterise the mobility of metals in the AMD and the extent of their immobilization, both in secondary Al and Fe precipitates and in a natural wetland downstream. Precipitates consisted primarily of jarosite with minor amounts of goethite, lepidocrocite, ferrhydrite and gypsum. A receiving wetland downstream acted as a sink for a number of metals, including Fe, Mn, Zn, Ni, Pb, U, Cu and Co, but residual Pb (1.3 mg L-1), Al (40 mg L-1), Fe (1.6 mg L-1) and Mn (6.5 mg L-1) concentrations persisted in the water downstream of the wetland. Abundant algal growth appeared as mats at AMD outwelling points and in the stream bed and on fringes of AMD retention ponds.

Rock dumps and slimes dams

Gold mine rock, sand and slimes dams cover an area of 80 square kilometres on the Witwatersrand. Many of these are no longer regarded as permanent structures but rather as an asset or resource as they are being re-worked for their gold and in some instances uranium and pyrite content. The waste from re-working is being re-deposited at residue sites governed by legislative constraints. Apart from the recovery of gold and uranium, about 70 to 75% of the pyrite is extracted. This limits the pollution potential of the new slimes dams (Kempe, 1983).

Sand and slimes dams affect the soil environment through wind and water-borne deposits derived from erosion, and through acid drainage. The texture of the tailings material is dominated by silt that makes it virtually cohesionless and highly susceptible to wind erosion. The steep slopes with high runoff, low infiltration, low water-holding capacity and extreme saline and acidic properties make slimes dams very unfavourable for natural plant growth (Van der Nest, 1994; Van der Nest & Coetzee, 1994; Beukes et al., 1998; Van Deventer & Hattingh, 2002). Despite their negative constitution, mine tailings can be converted into useful anthrosols. Nell (1997) investigated the agricultural potential and changes in soil hydrological properties resulting from the particular mining process led to a number of investigations (Schoeman, 2001).

De Villiers (1992a; 1992b) developed a system for classifying mine soils. Nell & Steenekamp (1998) investigated the properties of rehabilitated strip-mined soils and found that high bulk density, induced by soil handling and leveling operations using heavy equipment, was the most widespread and most limiting soil property. For example, the median soil bulk densities at nine different mines varied between 1.85 and 1.95 Mg m-3 at 200-600 mm soil depth. Acidification of the soils was found not to be widespread (Steenekamp & Nell, 1999).

Dumps and slimes dams, when removed by re-working, leave behind their compacted footprint on the underlying material. Those of slimes dams average 80 ha in size (Van Deventer, 2001). Although rehabilitation of slimes dams, rock dumps and their footprints may give rise to new sustainable ecosystems, Van Deventer (2001) warns that anthropogenic ecosystems such as these are sensitive and that rehabilitation results do not constitute walk-away solutions.

Aucamp (2000) investigated the trace element pollution of soils downslope of gold mine tailings dams at Machavi (North West Province) by wind and water erosion. The decomposition of sulphide minerals in the tailings resulted in pH values of 3.4 and the release of high, water-soluble concentrations of As, Cd, Co, Cr, Cu, Ni, Pb and Zn. These elements were determined in surrounding soils by means of NH4NO3 extraction. The total arsenic concentrations in tailings and soils suggest that As is a priority pollutant.

Lanyon et al. (1997a; 1997b) investigated the mobility of As from arsenopyritic rock and slimes dams at the Barbrook gold mine near Barberton in Mpumalanga. High As concentrations (up to 4.1 mg L-1) were mainly associated with bodies of standing water located on the tailings dams, characterized by pH values between 7 and 11. Arsenic mobility was linked to sorption by and co-precipitation with calcite. Red sesquioxide soil from the mining area was found to have a capacity to immobilize about 3500 mg kg-1 As.

Low-grade coal disposed on dumps amounts to 44.2 million tons annually. The total accumulated coal mine waste is estimated at 283 million tons and presently covers an area of approximately 1020 ha. The discarded coal contains pyrite as well as various toxic metals. The extent of soil pollution by metals resulting from stockpiling of discard coal has not been documented (Korentajer, 1992).

Soil degradation due to strip mining

Some of the most productive rainfed agricultural land in South Africa is found on the Mpumalanga highveld. Old, stable land surfaces with gently rolling topography, receiving 750-800 mm rainfall, are occupied by moderately deep, highly weathered soils with favourable water-holding characteristics and favourable response to fertilization. Strip (open-pit) mining became widespread in this area during the mid 1970s. Of a total of 2.7 million ha underlain by coal reserves in Mpumalanga, approximately 7% may ultimately be mined by strip mining methods (Limpitlaw et al., 1997). Guidelines for strip-mine rehabilitation were published (Chamber of Mines, 1981). A growing awareness of the serious loss of agricultural potential and changes in soil hydrological properties resulting from the particular mining process led to a number of investigations (Schoeman, 2001).
Prinsloo & Erasmus (2000) conducted maize field trials at three representative rehabilitated sites and found that while maize yields of approximately 5000 kg ha\(^{-1}\) is the norm for the vicinity, the average yield of four seasons, which included very good ones, was half of that at 2430 kg ha\(^{-1}\). At the 50th percentile, cumulative distribution functions of the average of all treatments indicated 1790, 2620 and 2880 kg ha\(^{-1}\), respectively, for the three sites. The economic break-even yield was 3100 kg ha\(^{-1}\). Maize production was thus proved uneconomical if soil limitations are not corrected. Schoeman et al. (2000) studied the soil water-holding characteristics at the above trial sites and found that water extraction by maize roots was strongly restricted to the upper 450 mm of the soil. This depth corresponded to that part of the profile that could be effectively reached by deep cultivation with conventional implements. Deeper soil layers (the deepest was 600 mm) were generally too dense at bulk densities of 1.75 Mg m\(^{-3}\) and higher for water to be effectively extracted. Evidence of hard-setting characteristics and some susceptibility to re-compaction were found. Schoeman et al. (in press), in quantifying the water balance on rehabilitated soils under pastures, found that high bulk density, coupled with hardsetting behaviour, is a widespread phenomenon in replaced cover soils, and can be rated as the number one problem affecting land use. This confirmed earlier findings of Tanner et al. (1986; 1987). Due to poor root distribution and shallow root development in places, caused by high bulk densities, particularly below 200 mm depth, and hardsetting behaviour during dry periods, much of the available soil water (at some sites, the bulk of it) is not extracted and utilized by the pasture crop, even during periods of high water demand. Under conditions of poor to moderate root development, the actual profile extractable water capacity (taking root distribution into account) can be as low as 20 to 30 mm, excluding water held at higher potentials than the drained upper limit. It was not implied, however, that rehabilitated soils with current low productivity cannot be made productive, as important basic ingredients of productive land exist such as moderate slopes, fair soil depth (Schoeman et al. 1997). Due to this wide variability, industrial effluents are complex because of the non-uniformity of most of their chemical properties. Many of the potentially harmful constituents present in industrial effluents cannot be easily biodegraded and are often toxic, even at very low concentrations. Wastes from chemical industries are very low concentrations. Wastes from chemical industries are very generally biodegradable while effluents from metal industries are not easily biodegraded and are often toxic, even at very low concentrations. Wastes from chemical industries are very complex because of the non-uniformity of most of their products. Soils generally have the capacity to sorb and inactivate many of the potentially harmful constituents present in effluents.

Environmental impacts on soils due to industries

Atmospheric deposition

The eastern highveld has become impacted by the acidifying effects of industrial pollution and by the intensification of agriculture and forestry. Held et al. (1996), Turner (1996) and Tyson et al. (1998) reported conditions in the atmosphere that were favourable for the formation of pollutants such as ozone and peroxyacyl nitrates, exhibiting phytotoxic properties. Sulphate and nitrate aerosols occurred that were the precursors of acid rain. Pollution trapped in the middle layer of the atmosphere is deposited by wet and dry processes. Wet deposits displayed levels of acidity similar to that of eastern North America and Europe. Extensive areas of sandy or loamy acidic soils occur that are reported to be prone to the effects of acid precipitation.

Singer et al. (1992) identified and reported white efflorescences (evaporites) coating grass vegetation during September in the Ermelo area and ascribed the phenomenon to atmospheric deposition during the winter months. It was postulated that deposition of 11-15 g m\(^{-2}\) a\(^{-1}\) of thenardite (Na\(_2\)SO\(_4\)), as was observed, must ultimately lead to considerable sodification of the soil and salinization of the groundwater.

Fey & Guy (1993) investigated the capacity of soils in the Vaal Dam catchment to retain sulphate, thereby modifying the salt load in runoff, and found it to be limited. Fey & Netch (1996) and Van der Merwe et al. (1989) reported extreme difficulties in detecting atmospherically induced salinization and sulphur additions to the soil above the natural background. Van Tienhoven et al. (1997) suggested that the classical linear relationship between total S and organic C in soil constitutes a useful basis for detecting the accumulation of S from atmospheric sources. Dodds & Fey (1997, 1998) and Fey & Dodds (1998) developed algorithms for classifying the sensitivity of South African soils to acidification. Topsoils from modal profiles were analyzed for acid neutralizing capacity (ANC) using a simple method of pH measurement in acetate buffer solution. The relationship between ANC and relevant textural and chemical properties was subsequently employed to classify, directly or by pedogenic inference, the series of the Binomial System into four sensitivity categories. Large areas of the eastern highveld are classified as moderately sensitive.

Industrial effluents and sludges

Industrial effluents can generally be classified into four broad groups (Kilani, 1993):

- effluents from food and drink industries, e.g. dairy;
- effluents from industries using animal or vegetable materials as raw materials, e.g. paper pulp mill and tannery;
- effluents from metal industries, e.g. iron and steel rolling mills;
- effluents from chemical industries, e.g. petroleum and pharmaceutical.

Due to this wide variability, industrial effluents are complex and difficult to treat. Waste water from the first two classes is generally biodegradable while effluents from metal industries are not easily biodegraded and are often toxic, even at very low concentrations. Wastes from chemical industries are very complex because of the non-uniformity of most of their products. Soils generally have the capacity to sorb and inactivate many of the potentially harmful constituents present in effluents.

SASOL (Ginster & Fey, 1994) investigated the feasibility of disposing effluent by irrigating grass (fescue) pastures, exploiting evaporation to get rid of excess water. The effluent used was nitrogenous and saline and contained potentially sufficient levels of fluorine and boron to inhibit plant growth.
Pre-treating the effluent with gypsum prior to irrigation was found to be effective for reducing the F concentration to tolerable levels and for improving the SAR of the effluent for irrigation purposes. It was found that plant B uptake was a function of the B concentration of the soil solution. Plant F uptake was strongly dependent on soil type. The F, B and salinity treatments appeared to influence the intensity of microbially mediated soil N transformations. Isolated patches of leaf scorch typical of B toxicity were observed but did not appear to have markedly lessened overall transpiration by the pasture cover. Soil quality of intensely irrigated sites improved markedly when allowed to lie fallow for a year. Groundwater quality was judged a more important criterion than soil properties for judging the stage at which the land treatment operation should be terminated or moved to a new site.

Thompson (1985; 1986) reported on the effects of the disposal of alkaline, high SAR (chloride) effluent from a pulp and paper mill in the Elands valley in Mpumalanga on kikuyu pastures growing on a deep clayey sesquisoildic soil of the Vimy series. After several years the only indications of soil deterioration were an increase in hardness (when dry) in the surface layer and a more pronounced angular blocky structure at depth. Rainfall simulator studies showed that the soil in question remained permeable at ESP values approaching 40, even at low TDS levels. It was concluded that markedly sesquisoildic clays, when subjected to saline and alkaline effluent disposal, are far less prone to damage than other soils with different clay mineral composition. Annually, large amounts of gypsum are used in ameliorating effluent from industries (e.g. paper mills and leather tanneries). Smith et al. (1992a; 1992b) pointed out that the use of industrial gypsum in ameliorating sodicity in soils and water leads to unacceptable F levels in soils, runoff and ground water infiltrates. Unacceptably high F levels were found at industrial water disposal sites where industrial gypsum was used at levels of 10-25 t ha\(^{-1}\).

Thompson (1981), whilst monitoring the effects of strongly alkaline sodic effluents on land, observed and subsequently confirmed experimentally that the presence of NH\(_4\) in the effluent resulted in an often markedly lower soil exchangeable sodium status than would be expected on the basis of the SAR of the effluent. Ströhmenger (2001) reported beneficial effects of a high NH\(_4\) supply under saline conditions and ascribed these to the antagonistic effect NH\(_4\) exerts on SO\(_4\) and Mg in plants.

McNab et al. (1995) reported on the monitoring of possible pollution effects of industrial waste water sludge applied to a dedicated land disposal site surrounded by sandy soils at Hammersdale in KwaZulu-Natal. The levels of mercury (originating from textile industries) in surrounding soils were of concern, but were found to be on the decrease from 29 mg Hg kg\(^{-1}\) in 1989/90 to current levels of less than 1 mg Hg kg\(^{-1}\) in topsoils, where it is bound in organic matter.

The disposal of industrial effluent by means of irrigation may have negative impacts on soil biota. Van der Merwe et al. (1993) used two microbial activity tests (dehydrogenase activity and nitrification) to study the effect of three industrial effluents from paper, petrochemical and hide industries on soil biology. In all the effluents, toxicity and salinity inhibited microbiological activity. In some effluents, this effect was temporarily masked to some extent by increases in organic C and pH, as these factors stimulate microbial activity.

Strydom et al. (1993) reported results of a postal survey on effluent production and disposal in the South African dairy industry. The majority of smaller factories and dairies dispose of their effluents by means of irrigation onto lands and pastures. Whether the exceptionally high chemical oxygen demand of milk and related products might have possible negative effects on soil biological activity was not indicated.

Hughes (1988) reviewed the use of tannery wastes in agriculture and singled Cr and Na out as the main pollutants. Hughes & Girdlestone (1989a; 1998b) investigated the composition of leather tannery sludge and its effects on sludge-applied soils (application rates of 0, 10 and 100 t ha\(^{-1}\)) packed in leaching columns. Some 70% of the air-dry mass of the sludge is proteinaceous organic material. Inorganic components include variable amounts of Ca, Mg, K, P, Na and Cr. The organic C, pH and CEC of the soil increased markedly, with the high Ca content of the sludge more than offsetting the sodicity. Chromium was not detected in any leachate. Hughes et al. (1995) reported on a pot experiment to determine the capacity of leather tannery sludges (some of them composted, alone and with pine bark) to supply nitrogen to wheat, taking potential sodicity problems into account. Plant growth, plant N and soil carbon benefited but sodicity affected growth severely in one of the sludges.

Hughes & Girdlestone (2001) reported three-fold increases in sugarcane growth on sandy soils to which large quantities of a combination of paper mill wastes and sugar bagasse were applied. The waste was almost entirely organic with a high C:N ratio, low levels of plant nutrients, very high water-holding capacity (almost 80%) and was considered neutral in its effects on soil salinity and sodicity.

**Peatlands and artificial wetlands**

The chemical purity of peat mined for use as a substrate for the mushroom industry is negatively affected by spills and discharges containing heavy metals. This is due to the remarkably strong capacity of peat to sorb pollutants from water. These may be transferred to the mushrooms grown on the peat substrate (J.P. Nell, personal communication).

The gravel beds used in artificial wetlands (Jansen van Vuuren & Coetzee, 1994; Van Schoor, 2000) may be considered anthrosols with a role to play in sustaining plants used in environmental health management.

**Industrial slags and organic wastes**

Finely ground metal industry by-products (slags) are regularly used as agricultural lime due to their acid neutralizing capacity. The slags are products of the process whereby ores are smelted with lime in order to extract impurities such as Si and several heavy metals and trace elements. These impurities can reach concentrations of several parts per thousand. With these slags, higher concentrations of Cr, Ni, Co, Cd, Cu, Zn, Mn, Mo, Ba and other elements are added to the soil than when natural agricultural limes are used.

Van der Waals & Claassens (2001) investigated in pot trials the plant availability of some of the abovementioned elements from slags originating in the Witbank area in Mpumalanga. The uptake of Co, Mn, Zn and Ba exhibited a
strong pH dependence, decreasing with increasing pH. Other elements exhibited various degrees of pH dependence and uptake. It was concluded that the use of slags as agricultural limes is safe within certain limits, as most of the heavy metals and trace elements are highly unavailable at the pH prevailing in soil after proper lime application. This could change, however, if the soil is left to acidify, particularly after years of slag application.

Hamilton et al. (1999) found abattoir waste to be an unsuitable substrate for pine bark-based potting mixes. Spent mushroom compost and sugarcane filter cake were found to be promising.

Fly ash
A major waste from industries is pulverized fly ash (PFA) deriving from coal-burning power generator plants. The 1990 production of PFA was estimated at 26 million tons. The largest fraction of the PFA waste consists of non-toxic inert materials, in particular silica and alumina. PFA may contain various hazardous components, in particular toxic elements, e.g. As, Ba, Bi, Mn, Cu and V. South African fly ash also contains high concentrations of B, which is phytotoxic. It does not contain pyrite and the pH of the material is very high (up to 12). Since heavy metal solubility and mobility generally decrease with increasing pH, the potential for heavy metal pollution from PFA dumps is very limited. On the other hand, boron may be leached out and contaminate water and soil resources (Korentajer, 1992).

The range between B deficiency and toxicity is remarkably narrow. The availability of B to plants is dependent on the B sorption characteristics of the soil. These are determined by pH (sorption increases with increased pH), clay mineralogy, organic matter, texture, exchangeable cation composition, ionic strength and water content of the soil (Bester, 1993). Bester & Fey (1992) pointed out that there is a need to establish permissible B loads for different soil types and plant species and concluded from a pot trial with Rensburg and Swartland soils, to which B was added, that relative yield plotted against B concentration in solution provides a useful differentiation of plant species in terms of B tolerance. Plant growth response could also be used as a bio-indicator of B hazard in soil environments when the difficult B analysis is not easily obtained.

Coal ash and phosphogypsum
Coal ash and phosphogypsum are major components of the chemical processing industry. Phosphogypsum is mainly derived from the treatment of phosphate rock with sulphuric acid. Phosphogypsum was thoroughly investigated as a potential soil ameliorant and fertilizer (McLaughlin, 1988; Frenkel & Fey, 1989; Dewey et al., 1989; Korentajer & Henry, 1989; Harmse, 1989; Stern & Laker, 1989; Korentajer et al., 1991). However, industrial phosphogypsum contains F, diminishing the product’s acceptability as a soil and water ameliorator. Smith et al. (1992a) reported unacceptable F levels in soils heavily irrigated with industrial water ameliorated with phosphogypsum. Although lower quantities were considered for agricultural use, the same trend may occur over time.

Bioreclamation
Du Plessis & Du Preez (1992), in an overview of the role of microbes in the reclamation of polluted soils, drew attention to the potential and significance of the use of microorganisms in degrading organic pollutants accidentally or purposely spilled onto soil. It was pointed out that organic pollutants basically contain halogenated aromatic or aliphatic compounds or combinations thereof. These are used by the microorganisms in the soil as carbon and/or nitrogen substrate. Variation in aliphatic chain lengths and substitution cause different compounds to differ in their resistance to bioremediation. Each unique set of pollutants, microorganisms and soil conditions has to be evaluated with respect to bioremediation. One of two approaches can be followed. In the first approach, it is assumed that the indigenous microbial community of the soil is capable of degrading the polluting compound. To stimulate these organisms, the factors impeding their growth are identified and ameliorated as far as possible. The second approach involves the introduction of inoculants (the so-called superbugs) to a contaminated site. Enrichment of the site with other utilisable substrates may be necessary for the introduced microorganisms to thrive.

Du Plessis et al. (1994) reported on the microbial catabolism of hydrocarbons in a petroleum and heavy-metal contaminated soil. They found sufficient indigenous microbial activity for hydrocarbon bioremediation under non-limiting conditions. Nutrient supplementation with N and P, together with aeration, is an important enhancing factor. Anaerobic conditions inhibited the process. Microbial catabolism induced low pH conditions in the soil, causing high heavy metal concentrations in the leachate. Adsorption of heavy-fraction hydrocarbons (>C20) to microorganisms and colloidal material was suspected of facilitating mobility of these fractions. The adsorption of microbes and pollutants onto specific soil surfaces (Du Plessis et al., 1994) and the porosity, particularly the intra-particle porosity of the medium (Du Plessis et al., 1998a; 1998b) affect the kinetics of attenuation.

Lees et al. (1995) reported on a pot and field trial dealing with the bioreclamation of soil contaminated with oil hydrocarbons at a disused refinery site. They followed the route of stimulating the indigenous microbial community by the addition of oxygen and nutrients. Soybean cultivars were screened for their suitability as bioreclamation agents under these conditions.

Environmental impacts on soils due to housing and urbanization

Sewage sludge
Sewage sludge is disposed of through land application, incineration and disposal into the sea. Of these options, land disposal is often the least expensive and is on the increase. Land disposal of sludges can supply crops with a large number of essential macro and micro-elements and organic matter. However, sludges also contain toxic substances in concentrations higher than those found in typical soils and may contain pathogenic organisms. Twelve major inorganic chemical contaminants (Cd, Cr, Cu, Pb, Ni, Zn, Mo, Hg, As, Se, B and F) were identified as being commonly present in sewage sludge (Smith & Vasiloudis, 1991). Sludges are applied to land in
one of two ways. In sacrificial or dedicated land disposal, high sludge loadings are applied to small land areas to reduce cost and environmental impact considerations (McLaughlin, 1984; Herselman, 2002). Over 55% of sludge in South Africa is disposed in this way. However, many of the sacrificial sites are nearing their life span and the pressure on the municipalities for environmentally sound disposal practice is increasing (Korentajer, 1991). In the re-utilization type of land disposal, the sludge is spread over a large area at lower rates and a crop is grown to try and recoup some of the nutrient value of the waste (McLaughlin, 1984; Steyn & Du Toit (1994); Snyman et al., 1998). This is being done by vegetable farmers at Philippini on the Cape Flats, in KwaZulu-Natal for pasture and sugarcane production (Korentajer, 1991) and in converting mine tailings into a growth medium (Van Deventer & Hattingh, 2002).

Korentajer et al. (1995) investigated the effects of levels of applied composted sewage sludge (0-4 kg m⁻²) on cabbage and Swiss chard in an irrigated field trial at Roodeplaat. In cabbage, concentrations and uptake of N, P, Cd and Zn increased with increased compost application level. In Swiss chard, concentrations of N, P and Cd increased, while Mn and Zn decreased. Heavy metals were preferentially accumulated in Swiss chard.

Herselman & Du Preez (2000) studied the bioavailability and leaching of trace elements under field conditions of sludge application (rates of 0-4 kg m⁻² were used) at Roodeplaat and on sacrificial land at Rooivale near Pretoria, using spinach and rye grass as indicator crops. As the plant uptake of all metals, except Cu, exceeded by far the maximum permissible levels for foodstuffs, it was concluded that high application rates of sewage sludge would definitely cause food chain contamination and, probably, toxicity in animals. Leaching of heavy metals was very limited.

Agassi et al. (1998) found in rainfall simulation studies that, contrary to expectations, sludge applied at 45 Mg ha⁻¹ reduced soil percolation rates, perhaps due to clogging of soil voids through microorganism proliferation and physical blocking by particles themselves. Some sludges display hydrophobic characteristics that reduce water infiltration (P.W. van Deventer, personal communication).

The maximum permissible concentrations (MPCs) of trace elements and heavy metals in sludges, soil and plants as well as the adsorption capacity and leaching profiles of sludge and heavy metal treated soils was researched in order to establish realistic norms (Steyn et al., 1994; Buys et al., 1998; Herselman, 1998; Herselman et al., 1998; Herselman, 2002). Van Zyl et al. (1999a) analyzed a large number of soils under grasslands for As, Cd, Cr, Cu, Pb, Hg and Ni in order to assess background values. The results for total, potentially bioavailable and immediately bioavailable content were compared with published MPCs. In many samples total contents exceeded the MPC (As, 37% of samples; Cr, 48%; Cu, 18% and Ni, 36%). It was concluded that the MPCs for arsenic and lead are too stringent, and that total content of trace elements is a poor indicator of soil pollution status and environmental risk. Research is needed to test MPCs for EDTA and \( \text{NH}_4\text{NO}_3 \) extractability, especially with respect to Cr, Cu and Ni. Herselman (2002) pointed out that current MPCs were set for unrestricted land use and that a need exists for a separate set of MPC guidelines for dedicated land disposal practices.

Snyman et al. (1999) concluded that none of the waste water treatment works in South Africa could comply with the Cu, Pb and Zn levels in sludge intended for unrestricted use. Henning et al. (2001), reporting on pot trials, found that the current heavy metal guidelines for soil metal concentrations were exceeded with respect to Pb, Cu and Zn, mostly due to high soil background levels. They concluded that the 1997 guidelines (WRC, 1997), when dealing with total metal content, set the standards for the unrestricted use of sludge on agricultural soils so conservatively that they cannot be attained within a reasonable framework of affordability and technology. Due to southern Africa’s diverse geology some soils indeed have naturally high background concentrations of trace elements. Cooper (1987), for instance, reported evidence of Cr(VI) toxicity to maize grown on soils with fluctuating water tables, derived from ultramafic rocks in Zimbabwe. Herselman & Du Preez (2000) found control levels of Cu and Pb at Roodeplaat exceeding the MPC for soil. In order to identify areas of potential toxicities, Herselman & Steyn (2001) embarked on a process of producing traced element maps by interpolating analytical data (determined by using a 0.5 M \( \text{NH}_4\text{EDTA} \) extract).

Approaches in the laboratory assessment of total elemental content and bio-availability were investigated and debated. While total soil analyses are universally used to set numerical limits for potentially toxic elements, they do not necessarily assess bioavailability. The latter is the key to understanding the behaviour and fate of elements in soils (Steyn & Du Preez, 1998; Du Preez et al., 1998; Herselman & Du Preez, 2000).

Alternative extractants for micronutrients were compared by Hammond & Beyers (1987). The effectiveness of different digestion methods vary (Kirsten & Steyn, 1998). Van Zyl et al. (1999b) compared results for As, Cd, Cr, Cu, Pb, Hg and Ni obtained with a number of extractants including EDTA (assumed to extract the long-term bioavailable fraction), \( 1 \text{M NH}_4\text{NO}_3 \) and 0.01 M \( \text{CaCl}_2 \) (assumed to extract the immediately bioavailable fraction) and \( \text{OAc-EDTA} \) (buffered to represent different pH conditions better) and found inconsistent ratios between results for specific elements using the different extractants. It can be concluded from their work that the use of a single extractant for all elements may not be the best route to take. Guidelines for sewage sludge analysis were published by Smith & Vasiloudis (1991).

**Phosphorus**

Due to increasingly stringent standards for sewage effluent P levels and improved P removal technologies, the P levels in sludge are increasing. The forms of P in sewage sludge are dependent on the composition of the raw sewage and the type of treatment. In conventional methods about one half of the P removed is settled out during primary sedimentation and one half in the activated sludge process. Before the secondary sludge can be applied to land it is generally subjected to a digestion (tertiary) treatment. These processes result in a mixture of soluble and insoluble organic and inorganic P compounds (McLaughlin, 1984). The P in sludges (2-4% on a dry solids basis) is receiving interest from the agricultural sector.
owing to the increasing cost of fertilizer P sources. Due to different forms of P resulting from different treatment processes, the plant availability is dependent on the processes in use. Generally, a large proportion is plant-available. Sewage sludges are generally not regarded as hazardous as far as P pollution of the environment is concerned. Although P is not leached from the soil as is N, it is recognized that P levels in sludge-amended land need to be monitored. It is recommended that, from a phytotoxicity viewpoint, sludge applications are reduced or ceased altogether when bicarbonate extractable P levels exceed 225 mg kg⁻¹ soil (Sabey, 1980, cited by McLaughlin, 1984).

Based on monitoring conducted during 1990/91, Van Niekerk (1992) concluded that 39% of the P and 47% of the N deriving from the Witbank Dam catchment each year was from municipal sewage treatment plants. The impact of these pollutants on irrigated soils would be largely beneficial.

Nitrogen
Sludge can be considered as a low-grade fertilizer, roughly equivalent to 4-2-0 or 5-1-0 commercial mixtures. In most cases the nitrogen content of the sludge determines maximum annual application rates. In principle, sludge is applied to provide plant nitrogen requirements equal to nitrogen fertilizer application rates. However, the availability of the sludge-derived N (and P) is difficult to predict. The organically bound nitrogen must be mineralized into inorganic forms prior to plant uptake. The generally low rate of release may be a desirable feature in situations where nitrate leaching is a factor as in sandy soil or in situations where volatilization losses of ammonia may be high, as in high pH soils. If application rates are too high, excess nitrate-N accumulates in the soil. As it is not adsorbed on the soil constituents, it may be leached below the root zone and may result in contamination of the aquifer. Such a situation was reported in the Cape Town area, where the nitrate N content of the groundwater was found to be greater than 20 mg L⁻¹ while standards for drinking water permit only 10 mg L⁻¹ (Korentajer, 1991).

Sludges as soil conditioners and ameliorants
Distinction is made between raw sludge (for which no agricultural use is currently permitted), digester sludge (use permitted for crops not eaten raw by humans) and irradiated or heat-treated sludge with respect to which the use is unrestricted (Oberholster, 1982).

Easton (1983) investigated the effects of anaerobically digested sludge containing a high industrial waste component on a sandy, nutrient deficient Clanshall soil with low CEC in a pot experiment, using Eragrostis curvula as test crop. Sludge application rates were equivalent to 0, 20, 50 and 100 dry solids per ha. The highest application rates resulted in yields 18 and 34 times higher than the control in the first and second harvests, respectively. However, the uptake of the heavy metals Ni and Cu, and the trace element Zn were also high (Ni and Cu near to or exceeding the upper critical limit and Zn considerably greater). Positive effects on the soil included a three-fold increase in time to wilting, a more than 50% increase in water-holding capacity, an increase in organic carbon from 0.2 to 2.0%, a reduction in bulk density from 1.43 to 1.20 g cm⁻³ and an increase in pH from 6.1 to 6.8. Zn, Ni, Pb and Cd were detected in the soil leachate. Pathogens in the sludge decayed within 5 months after application to below detectable limits for Ascaris. It was concluded that although there is no doubt that the use of the digested sludge improved the quality of the sandy, infertile soil, the use of such a highly industrialized sludge on agricultural land would require very strict control and careful management.

Champion (1985) investigated the effectiveness of dried sludge as P fertilizer on sesquisoic soils and found large responses in rye grass but also elevated metal contents. Preferential metal uptake and the effect of the presence of Zn on Cd absorption by the plant were pointed out.

Hughes et al. (1993) investigated the use of sewage sludge as a fertilizer and subsoil acidity ameliorant in a pot experiment. It was found that wheat yields increased greatly at high applications (40 Mg ha⁻¹) but that pH values were only raised at very high levels of application (160 Mg ha⁻¹), leading to nitrate leaching. Plant concentrations of N and P increased linearly with sludge applications up to 80 Mg ha⁻¹, followed by a leveling off at higher rates.

Solid household waste
Sanitary landfills
In urbanized areas, solid household waste, generated at a rate of 1 to 2 kg per person per day (Malan, 1987) and trade waste ends up in sanitary landfills where refuse is deposited, compacted, covered with soil layers and left to decompose under relatively controlled conditions and within a legal framework (Van Niekerk, 1994). The process is dependent on the availability of suitable soil resources, and impacts on these by taking up land space, borrowing cover soil, changing the properties of the cover soil and resulting in areas of man-made soils with temporarily limited land use options as an end result. The decomposition goes through four stages, each with a characteristic leachate composition (Ross, 1990). The first is an aerobic stage during which carbon dioxide is produced. This is followed by a longer anaerobic or fermentation period, the early stages of which (second phase) sees a decline in leachate pH, mobilization of metals and the accumulation of fatty acids and alcohols. During the third phase, the simple acids and alcohols formed are used by methane-producing bacteria. Metal concentrations, chemical oxygen demand and acidity of the leachate are thereby reduced. The fourth, stable methanogenic stage is reached when the methane and carbon dioxide proportions in the gas are almost equal and remain constant (Röhrs et al., 1998).

Although soil coverings (intermediate and final) constitute only a small fraction of the landfill by volume, the total surface area per unit volume of the soil greatly exceeds that of refuse. Soil differentially adsorbs organic and inorganic compounds and microorganisms and due to its very great buffering capacity, protects the landfill bioreactor against surges of certain chemical or even physical changes (Du Plessis & Hughes, 1995).

Co-disposal of solid household waste and sewage sludge was found to be beneficial as it enhances the rate at which refuse degrades. The acid-producing phase is shortened and gas production is concentrated over a shorter period, making
the extraction of methane more viable and settlement of the waste bulk is sped up (Röhrs et al., 1998).

**Composting**

Under less formalized housing conditions, organized household waste disposal may be lacking or irregular. Under these circumstances, composting with added N or manure may be advantageous to environmental health and urban soil use.

Mubyana & Korentajer (1998) investigated local fungal species for their ability to aid the composting of domestic solid waste and proposed a number of species for the purpose.

Korentajer et al. (1999; 2001; 2002) developed methodologies and design criteria for the composting of urban waste at household and community levels. They found that urban agricultural land use may benefit from the composting of household waste, but that vegetable discard that is beneficial to composting is in short supply. All mature composts (C:N ratios of around 11) were found to be likely to have similar agronomic value. They reported no environmentally harmful effects of the use of household waste compost. They pointed out that the agronomic value is determined by the C:N ratios of the raw materials and that increased rates of application of low quality organic waste materials (C:N ratios >20) did not result in increased vegetable yields. Using such low-cost, slow release forms of N in urban agriculture is environmentally friendly in that they result in little or no leaching of N under the poor irrigation management conditions which commonly apply. Graham et al. (1998) reported on the nutrients and organic substances that leach from compost heaps. Fluxes vary with various stages of the composting process. Nitrate was reported to increase sharply after about 45 days.

As latent phytotoxicity may persist in visually mature composts, Smith & Hughes (2001) proposed a method for determining compost maturity.

**Urban runoff**

Surface runoff from urban areas and roads may contain, inter alia, heavy metals such as Zn and Pb. Reid et al. (1997) found that the sediments in Zoor Vlei near Cape Town acted as a strong sink for these elements. Sorption reactions are involved. Acidification of the water in equilibrium with the sediments may cause the release of pollutants.

**Environmental impacts on soils due to fire and natural disasters**

Fires, controlled or uncontrolled, exert an environmental influence on the soil. Jones et al. (1990), in a study to determine which soil organic and biological compounds are affected by fire regimes, found that organic C, total N and N mineralization potential decreased with increasing fire frequency. The ratio of microbial biomass to total soil organic C in the top 150 mm was much higher in unburned plots than in burned plots (0.008% compared to 0.004%).

Materachera et al. (1998) studied the influence on selected soil components of frequency of burning the grass cover. Depending on the quantities of biomass being burned, the elements deposited in the ash during burning tended to enrich topsoils in inorganic nutrients (mainly Ca, Mg, Na, K), resulting in a rise in pH. These elements were stated to be highly soluble and are readily leached from the soil surface. Regular fires also resulted in reduced soil organic C and extractable P in the topsoil. This was ascribed to reduced biomass addition, and possibly lower microbial activity in the rhizosphere.

In a case study on the effects of a wildfire on a forested catchment in the southwestern Cape, Scott & Van Wyk (1990) found that the wettability of the soils was greatly reduced. Surface soil layers (0-10 mm) were burned clean of any inherent water repellency, but more severe repellency, in broader bands, was induced in deeper soil levels. It was postulated that the widespread development of water repellency led to overland flow during larger rainstorms, high soil losses and marked changes in the hydrological behaviour of the catchment relative to the unburned condition.

**Environmental associations between soils and nutrition-related disorders in animals and humans**

Laker et al. (1981) reported on four systematic soil studies that revealed consistent associations between soil factors and the incidence rate of oesophageal cancer in less developed rural areas. An integrated model describing the relationships was presented. The main theme of the model is that the cancer incidence rate is related to the level of some mineral element in humans or in the crops which form their staple diet; that an abnormal level of such an element (deficiency or toxicity) may cause physiological abnormalities in the human or the staple food crop, which leads to the production of carcinogenic substances; and that the mineral element level in the staple food depends upon the level of availability of that element in the soil, which, in turn, is dependent on various soil factors, especially soil pH; these soil factors are determined by various environmental factors.

Gummow et al. (1992) investigated illthrift problems experienced in cattle in the vicinity of a vanadium plant in the Middelburg district, Mpumalanga. Abundant circumstantial evidence was accumulated that led the authors to believe that animals were suffering and dying from the effects of excessive levels of airborne vanadium pentoxide deposited on pastures and soils. Background values of V in topsoils, for example, ranged between 1 and 7 mg kg⁻¹ (NH₄-EDTA extraction; Gummow et al., 1992; Kaempfier et al., 1993) whereas values immediately downwind of the vanadium plant ranged between 300 and 350 mg kg⁻¹. It was suggested that the pollutant entered the food chain via direct deposition on pastures as well as being taken up by plant roots. Due to the vague nature of the symptoms of vanadium toxicity and a lack of definitive diagnostic tools for vanadium related diseases, it could not, however, be conclusively proven that animals had died or suffered from vanadium toxicity.

Neser et al. (1997) and Smith & Neser (1998) reported on elemental imbalances in the soil, plants and animals of the Ghaap Plateau (Northern Cape province), leading to a condition in calves called Vryburg hepatosis. These imbalances were caused by excess soil manganese, inducing deficiencies in Fe and Co. Affected areas showed high Mn to Fe ratios in the soil. These are transferred to plants and to calves eating the soil in an effort to correct imbalances in their diet. It was found that Co is adsorbed in the Mn/Fe concretions, resulting in low available soil Co and low Co levels in the vegetation. As Co is an essential part of vitamin B12 in animals, the com-
petition of Mn with Co also resulted in vitamin B12 deficiencies in calves.

Pooley et al. (1997) and Cerutti et al. (2003) linked unusually high incidences of dwarfism and the endemic occurrence of Mseleni Joint Disease (MJD) in a narrow north-south corridor of the Maputaland Coastal Plain to nutrient deficiencies in the recent Quaternary sands (Fernwood soil form). Soil samples were collected along transects through the high incidence area. Pooley et al. (1997) found a sub-optimal supply of Ca, P, Zn, Cu and B, and as all the deficient elements have been associated in medical literature with skeletal disorders, hypothesized that these might exert their influence synergistically. Cerutti et al. (2003) confirmed all soils to be deficient in Bray-I extractable P and ammonium-EDTA extractable Cu and Zn, with respect to critical levels for maize growth, having averages of 4.5, 0.5 and 0.4 mg kg⁻¹, respectively. There was a marked difference in ammonium-acetate extractable K and ammonium-EDTA extractable Se between the low and high incidence areas, with average values of 209 and 27 mg K kg⁻¹, and 0.46 and 0.09 mg Se kg⁻¹, respectively. Other nutrients studied (Mg, Ca, K, Mn, Co, V, Mo & B) did not show anomalies between the two areas.

In a subsequent study (Cerutti et al., 2003), topsoil samples were collected at 1 km intervals along a 34 km roughly east-west transect through the MJD high incidence area. These were analyzed for P, K, Mn, Fe, Cu, and Zn (Ambic-2 method), Ca and Mg (KCl extraction) and B (hot water extraction). In a subtractive maize growth pot trial, using a complete nutrient solution from which one element was withheld per treatment, yields for the minus P, K, Ca, S, and Zn treatments were all below 80%, relative to the complete treatment, indicating deficiencies of these elements. Plant tissue analysis showed deficiencies of P, K, Ca, Mg, Cu, and Zn. Treatments within the landscape of multiple deficiencies were indicated, with Cu and Zn deficiencies throughout the landscape.

McCaflrey & Willis (1997) reported elevated fluoride levels with respect to most areas surrounding the Pilanesberg Igneous Complex and the areas underlain by granitic rock of the Bushveld Igneous Complex in North West Province. The source and mobility of F⁻ in the hydro-geochemical evolution of the area was established, as well as the spatial relations between F⁻ enriched groundwaters and fluoride sources in rocks, mineralized areas and soils. A map of areas of zero, medium, high and very high fluorosis risk was compiled. In some parts F⁻ levels exceeded S.A. guidelines by 800%.

Soil quality indicators

During South Africa's years of relative international isolation, the World Bank in collaboration with various international environmental agencies progressed in developing the concept of land and soil quality indicators. Frameworks of principles were developed within which local countries were to develop their own sets of land and soil quality indicators. Ideally, land and soil quality indicators should be limited in number so that main trends are highlighted, but at the same time the issues should not be oversimplified. The Department of Environmental Affairs and Tourism (2001) initiated an ongoing process of developing a core set of environmental indicators for state of the environment reporting.

Nell et al. (2000) and Nell (2001) proposed and applied a set of soil quality indicators for the Mpumalanga Province. Data were collected for soil pH (water), organic C, N, P, Ca, Mg, Na, SAR and EC. These were stratified according to a number of vegetation/land use groups (grassland, savanna, cultivation for maize production, cultivation for other crops, and rehabilitated coal strip mine soils). It was found that soil qualities under pressure are mainly those associated with soil organic matter (pressures are monocultural cultivation and mine rehabilitation standards). Nutrient depletion under commercial agriculture (as indicated by N, P and pH) was not identified as being under serious pressure (the median N, P and pH values of topsoils used for maize were 0.08%, 17.7 mg kg⁻¹ and 5.89, respectively).

Ströhmenger & Beukes (2002) found a strong relationship between soil P concentration and the intensity of degradation of natural soil plant cover in Gauteng. P concentrations were found to increase with increasing degradation intensity. This phenomenon was ascribed to lessened P uptake by degraded vegetation and an increased surface albedo causing higher P mineralization rates.

The way forward

Sustainable use of land and soil to the benefit of society and the environment is described by Blum (2002) as harmonization of various land uses and the competition between them, and the exclusion or minimization of those land uses that result in irreversible damage. (Impacts that cannot be stabilised or reduced by natural processes or by human interference within 100 years should be considered as irreversible). Harmonization of agricultural production and the environment is not primarily a scientific, but rather a political target. Science can only contribute through the development of scenarios. Decisions are taken or influenced by politics.

The divide between scientific and technological knowledge and technical ways of presenting information on the one hand, and politics and decision making on the other needs to be bridged. Technical data are to be presented with decision makers in mind. The information that exists in the country does not necessarily tell what needs to be known about the environment and whether sustainable development is being achieved or not. Most data informs about a particular condition, but not necessarily about impacts. The single most important message that comes through from the 1999 national State of the Environment report, it is that we require improved data and information on the impacts of our activities on the environment. (Department of Environmental Affairs, 2001).

In dealing with challenges for soil science in the 21st century, Blum (2002) not only envisages scientific-technical challenges, but cultural, societal and economic challenges in particular. Markets provide a challenge by preferring products produced or manufactured in an environmentally friendly way. International conventions challenge government and society to work towards environmental sustainability. They also challenge scientists to package results so as to have a political cutting edge.

In preparation for the World Summit on Sustainable Development (WSSD) in Johannesburg, DEAT published an introspective review of the ten years of progress since Rio
The most pressing priorities facing South Africa were identified and recommendations put forward. The following are of particular interest to soil science: DEAT has been urged to take the lead in development of a national monitoring and evaluation system and indicators for sustainable development; the workability and implementation of the National Environmental Management Act (NEMA) principles is to be improved at local government level, as well as the linking of national and provincial planning to local levels; funding agencies, science councils and tertiary institutions are urged to invest in further research and development (R&D) on environmental health issues; incentives are to be provided for the accumulation of natural capital (via environmentally sound production and consumption practices); the interlinking of components within the governance framework is to be improved and key policies are to be harmonized at all levels; R&D pilot projects on alternative water resources such as wastewater recycling are to be intensified.

The scope of potential R&D in soil-environmental issues is indeed wide. If priorities were to be narrowed down to five, the following might perhaps be singled out:

1. The need for a suitable national environmental monitoring and evaluation system. In its absence, various ad hoc pieces of environmental data, information, norms and standards that are constantly being collected will remain pieces of a large puzzle that is extremely difficult to incorporate into a holistic picture, e.g. for state of the environment reporting.

2. The need for improved cross-linking and synergy between various sectors impacting on the environment (and their governance). For example, Minerals and Energy "borrows" land from agriculture, mines the minerals and returns the land after closure. Current policies, despite the level of consultation between the two sectors, often lack effectiveness.

3. Soil science needs to be recognized as a full partner by DEAT.

4. In the current political climate of maximum devolution of decision-making, the local government level is critical to environmental management. Clear principles, empowerment of officials and compliance with legislation will be increasingly important at this level.

5. The safe and sustainable use of sewage sludge/ fly ash combinations has the potential to greatly improve the physical (water-related), chemical and plant nutritional properties of agricultural soils. This may perhaps be viewed as the single most promising future avenue of environmental R&D.

References


Africa. 4th Int. Symposium on Environmental Geochemistry, Vail, Colorado.


SCHOEMANN, J.L., KRUGER, M.M. & LOOCK, A.H., 1997. Water-holding capacity of rock fragments in rehabilitated open-


VAN DEVENTER, A.J., SCHOEMAN, J.L., KORENTAJER, L. 
on soils of the Eastern Transvaal Highveld and adjacent areas 
in the Republic of South Africa. Report No. GB/A/89/14, 
ARC-Institute for Soil, Climate and Water, Pretoria.

VAN DER MERWE, P.J.G., NEL, E., SMITH, H.J.C. & VAN 
WVK, A., 1993. Evaluering van die effek van industriële uitvloe-
isels op grondbiodegradasie en ontkieming met behulp van 
mirobiologiese aktiwiteitsbepalings en ontkiemingstudies. S. 

VAN DER NEST, L.J., 1994. Rehabilitation of gold tailings dams - 
A new approach. Seminar on Sustainable Land Use, 13-14 Sep-
tember, Pilanesberg, 157-167.

goudmyndilik vir die vestiging van plants. 18th Nat. Cong. 

availability of selected heavy metals and phosphorus from 

Die invloed van ‘n steenkool-afgelei fulviensuur op koper en 
nikkel toksisiteit vir die myndilik. 18th Nat Cong. Soil 

VAN DEVENTER, P.W., 2000. Die invloed van uitruibare natri-
umpsentasie en kleimineralogie op die infiltreerbaarheid van 
gronde wat reeds as gevolg van sikliese besproeiing verseil is. 
WRC Report No. 499/1/00, Water Research Commission, Pre-
toria.

VAN DEVENTER, P.W., 2001. Future land use of man-made land-

ification system for South African Anthrosols. Joint Cong. Soil 

on the rehabilitation research project conducted at FS S 6 slimes 
dam for Anglo Gold. Envirogreen, Potchefstroom.

VAN NIEKERK, A.M., 1992. Development of water quality man-
agement plan for the upper-Olifants River basin. Water week 

VAN NIEKERK, A.M., 1994. Environmental criteria for waste dis-
posal site selection. Seminar on Sustainable Land Use, 13-14 
September, Pilanesberg, 168-182.

VAN SCHOOIR, L., 2000. The use of artificial wetlands in the puri-
22 August 2002.

VAN TIENHOVEN, M., DODDS, H.A., FEY, M.V. & WILLIS, 
J.P., 1997. Baseline survey of air pollution impacts on soil and 
water quality in Mpuimalanga province, South Africa. 4th Int. 
Symposium on Environmental Geochemistry. Vail, Colorado.

trace element concentrations for South African soils. 22nd Cong. 

VAN ZYL, A.J., DU PREEZ, H.G., LAZENBY, E. & KIRSTEN, 
W.F.A., 1999b. Evaluation of extraction methods estimating 
trace element bioavailability in South African soils. 22nd Cong. 


sludge (1st edn.), Water Research Commission, Pretoria.