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Environmental impacts associated with an abandoned mine in the Witbank Coalfield, South Africa

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Abstract

Mining at Middelburg Colliery in the Witbank Coalfield commenced at the turn of the last century. Initially, there was little environmental degradation associated with mining activities; however, in the late 1930s, a pillar-robbing programme commenced. This has had a marked effect on the environment. Some of the most notable primary effects include subsidence, the appearance of tension cracks at the surface and crownhole development. Secondary effects include spontaneous combustion of the coal worked, as air has been provided with ready access to the mine, accelerated subsidence due to the strength of many pillars being reduced by burning, and a marked deterioration of groundwater quality in the area due to the seepage of acid mine drainage from the mine. Spoil heaps also form blemishes on the landscape. These contain significant amounts of coal and have undergone spontaneous combustion. The deterioration in the quality of water has led to the decimation of vegetation in some areas and the eradication of aquatic flora and fauna in a nearby stream. © 2001 Elsevier Science B.V. All rights reserved.

Keywords: pillar failure; void migration; spontaneous combustion; spoil heaps; acid mine drainage

1. Introduction

Coal mining unfortunately tends to make a notable impact on the environment, the impacts varying in severity depending on whether the mine is working or abandoned, the mining methods used, and the geological conditions. This paper surveys the environmental degradation associated with the abandoned Middelburg Colliery in the Witbank Coalfield, South Africa (Fig. 1).

Mining operations at the mine concerned began in 1908, the coal being worked by the bord and pillar method. In the late 1930s and early 1940s, pillars were robbed over an extensive area of the mine. This resulted in the formation of crownholes at the surface due to void migration as the collapse of bord areas occurred, and of discontinuous subsidence caused by multiple pillar failure. The subsidence resulted in extensive surface fracturing above the workings. In 1947, when the mine was being decommissioned, spontaneous combustion within the work-

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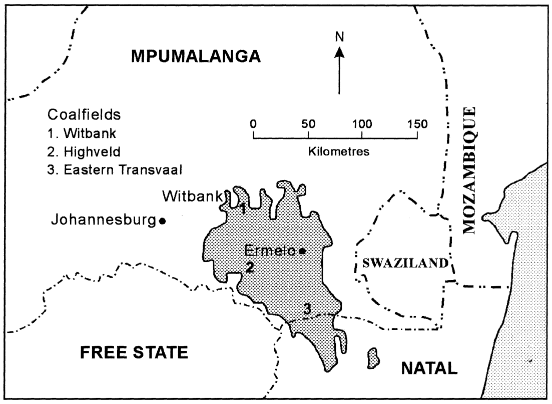

Fig. 1. Location of Witbank Coalfield.

ings was noticed for the first time. Despite efforts to extinguish the fire, it is still burning, emitting noxious NO_x and SO_x into the atmosphere. The subsidence has had an impact on the groundwater. For example, recharge into the old workings from rainwater has been enhanced by extensive fracturing of the ground surface and the appearance of crownholes so that now more than 50% of the rain which falls infiltrates the ground over most of the mined area. In the east of the mine property, the groundwater issues from the mine in a number of small springs. This water, which is highly polluted, makes it way into a local stream. The polluted groundwater has a high total dissolved solids (TDS) concentration and a low pH value. As a consequence, plant life over an area of 3 ha, where the seepage occurs, has been decimated.

2. Geology and mining history

The coal seams in the Witbank Coalfield were formed in an epicontinental environment and occur within the Vryheid Formation. The Vryheid Formation forms the midpart of the Ecca Group, which, in turn, is part of the Karoo Supergroup. This formation consists of sediments deposited in shallow marine and fluvio–deltaic environments in which coal developed from peat which accumulated in swamps and marshes. The formation primarily consists of sandstones, siltstones, mudstones and shales. As the northern margin of the coalfield is approached, the sediments thin and the Vryheid Formation rests unconformably on the basement rocks, i.e. the Transvaal Supergroup, the Waterberg Group, and volcanic rocks associated with the Bushveld Igneous Complex.

The five recognized coal seams in the Witbank area are named, consecutively, 1 to 5, the latter

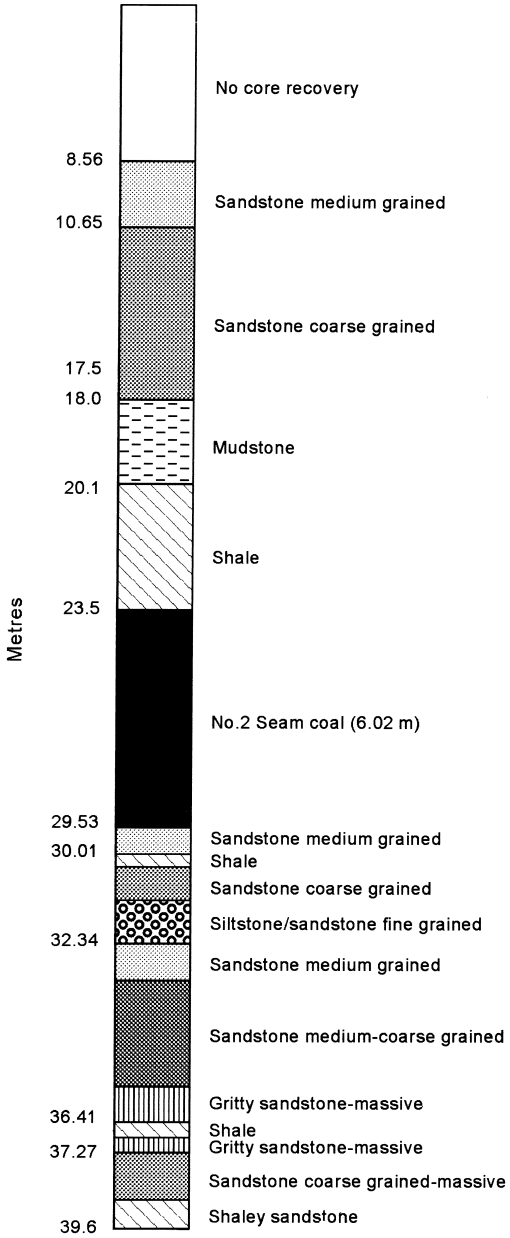

Fig. 2. Drillhole log showing the sequence of strata in relation to no. 2 seam, Middelburg Colliery.

being the youngest. These seams occur within a succession some 70 m in thickness. At the mine, the no. 1 seam is poorly developed. The no. 2 seam, which was the only seam mined during the life of the mine, is found at a depth of approximately 18–23 m and ranges in thickness from 3.5 to 6 m (Fig. 2). A SSW–NNE-trending anticlinal axis is present in the area. As a result, the coal seam dips to the west and to the east, on either side of the axis. The dip of this coal seam rarely exceeds 5°. In the east of the mine property, the no. 2 seam crops out approximately 100 m west of a local stream, namely, the Blesbokspruit. Seam nos. 3–5 have been removed in this area by post-Karoo erosion.

The unweathered strata overlying the no. 2 seam consists of shale, fine-grained sandstone, interbedded sandstone and siltstone, and mudstone, and has an average thickness of 10–15 m (Fig. 2). The weathered zone varies in thickness, ranging between 5 and 10 m, and consists of weathered sandstones and shales with an abundance of ferricrete.

No. 2 seam was exploited using four different mining methods. These were bord and pillar, pillar robbing, stooping, and opencast mining. Fig. 3 shows the geographical distribution of the different mining methods within the mine property.

Bord and pillar mining started in 1908. The average pillar size left behind was 6×6 m and the average bord width was 7 m, so the extraction ratio was approximately 60%. The average mining depth was in the region of 18–20 m, with a mining height of 2.5 m. From 1908 to 1947, when operations at the mine ceased, a total area of some 1700 ha was undermined using the bord and pillar mining method.

Pillar robbing started in the late 1930s and continued until 1946. The original pillars were quartered, leaving four smaller pillars at each corner. The remnant pillars comprised approximately 25% of the original pillar. This meant that the extraction ratio increased from 60% to about 90%. Any smaller pillars (4×4 m) were robbed on two sides, leaving approximately 33% of the original pillar intact. Pillar robbing occurred over an area of approximately 215 ha.

Stooping is a technique that entails removing 85% of the original pillar. On backward retreat, the hanging wall caves into the goaf. Stooping was used in 1982 and 1983 in an area of 6 ha and was to ensure the long-term stability of a local municipal road, which was to be constructed, from subsidence damage and to protect it from the underground burning coal.

Opencast mining of the boundary pillar of the mine began in 1991 and ceased in 1992. This operation was undertaken to prevent the spread of the underground fire into adjacent mines. In addition, a number of small coal remnants were mined by opencast method, notably a suboutcrop boundary pillar in the east of the mine property.

3. Subsidence

In the pillar and bord method of mining, pillars of coal are left in place to support the roof. The pillars, therefore, have to sustain the redistributed load attributable to the overburden, which means that the strata immediately above and below the workings are subjected to added compression (Bell, 1988). Stress concentrations tend to be located at the edges of pillars and intervening roof beds tend to sag (Wardell and Wood, 1965). Surface subsidence may be an expression of either multiple pillar failure or bord collapse with accompanying void migration.

Slow deterioration and failure of pillars may take place after mining operations have ceased. This is particularly the case if pillars are robbed on retreat, that is, as the mine is coming to an end of its working life. Obviously, the stress on a pillar increases as the extraction ratio increases.

The roof rock in the bords may collapse with time. This leads to void migration but the rocks that collapse bulk, so that void migration ultimately comes to an end. However, if seams are at shallow depth (e.g. less than about 10 times the height of the workings), then void migration can give rise to the appearance of crownholes at the surface.

There is little evidence which suggests that the original mining method caused subsidence in the area. This is understandable since mines with extraction ratios of up to 70% often are relatively stable (Bell and De Bruyn, 1999) and, as remarked, the extraction ratio was around 60%. Subsidence originated in the late 1930s when pillar robbing was practised, the extraction ratio being increased substantially. Pillar robbing increased the load which the

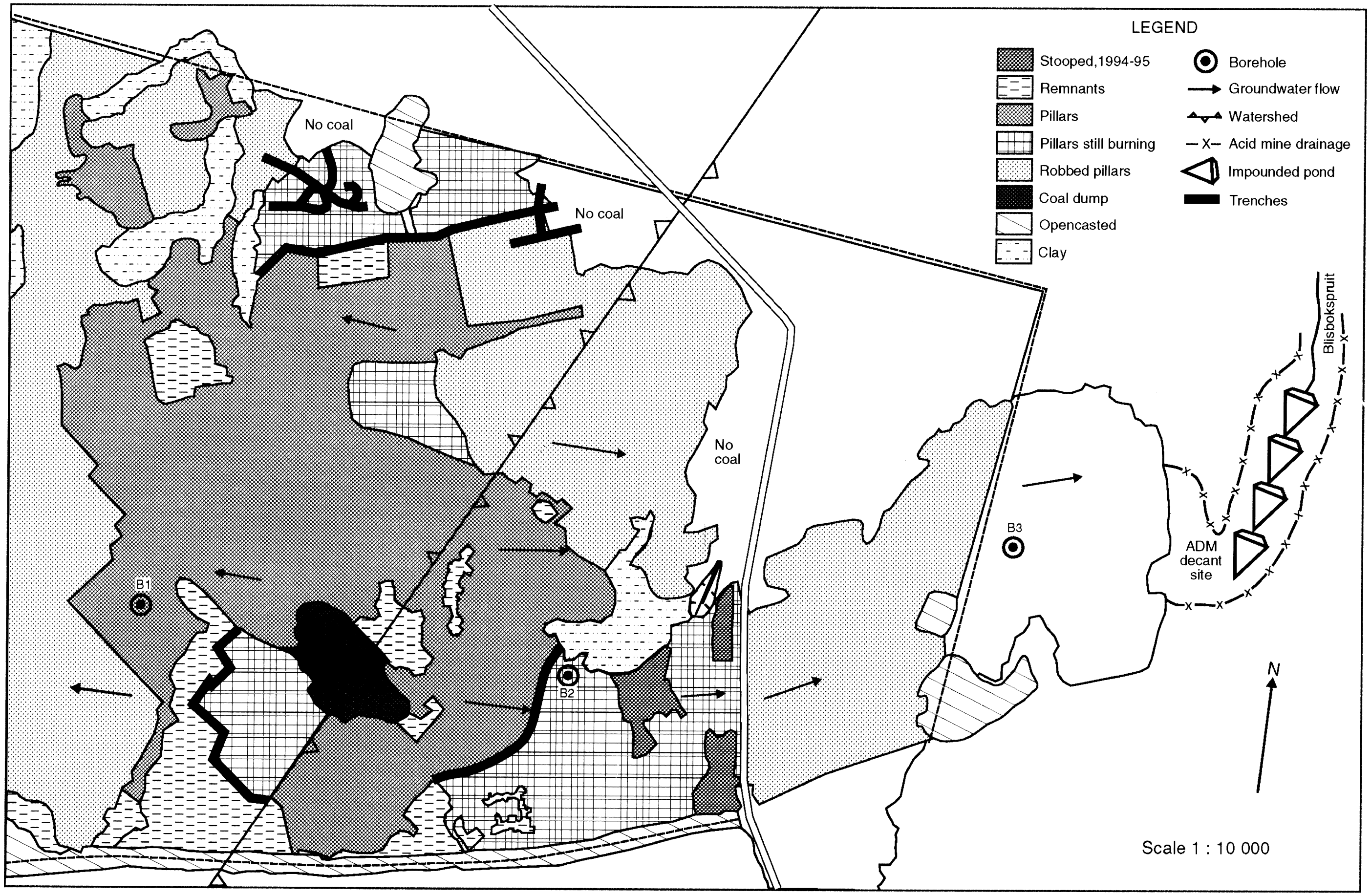

Fig. 3. Mining methods used at Middelburg Colliery. Also shown are areas where underground fires are still burning, the local watershed and the area of acid mine drainage. Note that one of the spoil heaps is just off the map to the immediate west.

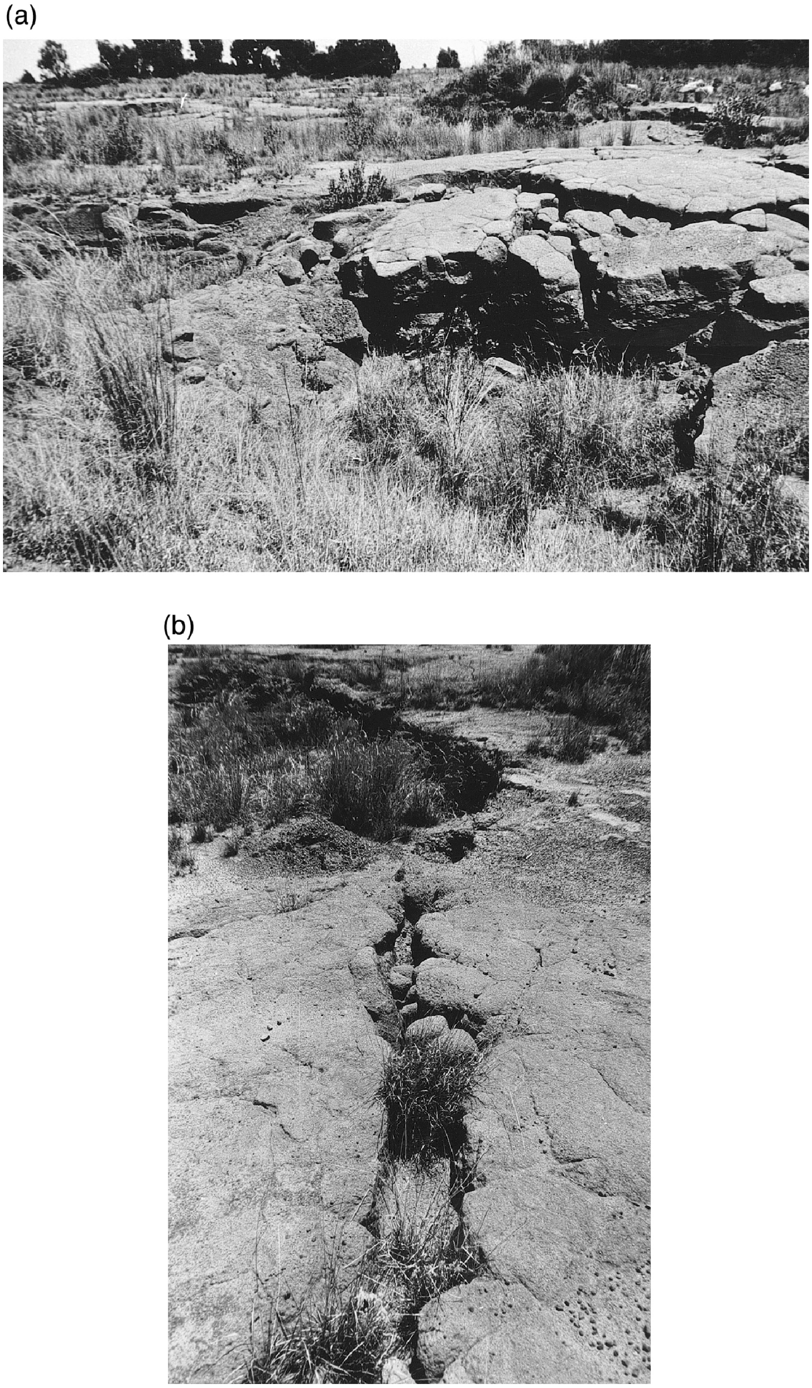

Fig. 4. (a) Subsidence due to pillar failure. (b) Tension fracture developed in sandstone running away from an area of subsidence due to pillar failure.

pillars had to carry by as much as one-third. Robbing also changed the shape of pillars by reducing their width significantly while the pillar height remained constant. This resulted in a reduction in pillar strength to the extent that many pillars could no longer support the overburden stress, with failure occurring when the ratio of pillar strength to vertical stress becomes less than one. The failure of a single pillar increases the stress on surrounding pillars, causing them to fail in domino fashion. The surface subsidences caused by multiple pillar failure are generally several hundred square metres in extent, and the collapsed areas often are bounded by near vertical sides (Fig. 4a). Surface tension cracks around the outer edges of the collapsed areas are typically 200–800 mm in width, and can extend in length for up to 100 m (Fig. 4b).

The roof strata at the mine are generally weak and incompetent shales, and are in tension due to flexure between the pillars. Obviously, the length of the span between pillars and the strength of the rock beam are all important in terms of roof collapse. High compressive stresses acting in the vicinity of the pillars can cause failure of roof strata. Stresses tend to develop at an angle to the maximum compressive stress (Piggott and Eynon, 1978). In the incompetent roof strata, this tends to be at a high angle (75°). These stress fractures, combined with interpillar tensional areas, have caused collapse, resulting in upward void migration through overlying strata, until the weathered zone is reached. Generally, the weathered material has subsided by 2–3 m, but in some cases, the weathered material has collapsed totally into the old workings, leaving voids 15–20 m deep. The resultant crownholes at the surface have diameters between 5 and 10 m (Fig. 5). Tension cracks, varying in width from 200 to 500 mm, are found around the perimeter of the larger crownholes.

Subsidence has led to flooding in places (Fig. 6). However, the impact of subsidence on property has been minimal as, fortunately, there is no major infrastructure in the area. Nonetheless, in the early 1980s, a municipal road leading to the local airport was constructed over the mine property. Stabilization of the area prior to road construction was necessary due to the high risk of surface subsidence and the spread of the underground fire. This was achieved by stooping the remaining pillars lying below the western side of the road, thus causing total collapse of the underground workings. Directly beneath and on the eastern side of the road, the bord areas were filled with a sand–cement grout mixture to provide additional support for the road foundation. These measures have been successful, as the road has not been affected by the subsidence, which is common in the surrounding area.

Fig. 5. Development of a large crownhole right of centre with curved tension cracks developed to the left.

Fig. 6. Flooding due to subsidence.

One of the few subsidence incidents to affect structures occurred in 1991 when a high-voltage power line pylon was affected by crownhole subsidence. The crownhole appeared after a period of heavy rain, directly beneath two limbs of the pylon. Remedial action included the placement of two horizontal struts beneath the pylon in order to spread the load onto more stable ground, together with the installation of a pulley system on the pylon to allow overhead cable movement in the event of further subsidence.

4. Spontaneous combustion of the coal seam

All entries to the mine were sealed on closure in 1947. However, a shaft was opened for access when an area was mined by stooping in 1982–1983. This was sealed when stooping ceased. Hence, except for this limited period, air and water have not gained access to the mine by mine entries. The coal in the mine has been undergoing spontaneous combustion for over 50 years and it is estimated that the area affected by burning is between 150 and 200 ha. The burning area is some 2 km from the nearest town.

The characteristics that affect the susceptibility of a particular coal to self-heating include temperature, rank, surface area exposed, moisture content, and pyrite content. Obviously, the rate of self-heating increases as the temperature increases, and so, once ignited, the burning process can be self-sustaining, if there is a continuous supply of oxygen. The tendency of coal to self-heat increases with decreasing rank of coal. According to Michalski et al. (1990), as rank decreases, the seam moisture content, oxygen content, internal surface area, and air permeation tend to increase. An increase in the natural moisture content of coal can liberate heat, and greater surface area and air permeation have the same effect. If the pyrite content of coal exceeds 2%, then, this also aids the self-heating process as oxidation of pyrite also is an exothermic reation.

Mining creates pathways through which air can be carried to coal. However, the retention of heat by the coal is largely dependent on the air flow, in such a way that there is a critical velocity (which varies with the other factors mentioned) below which the coal is oxidized but the air flow is not capable of removing the heat generated. Such conditions commonly exist in partially collapsed mines. Furthermore, in old abandoned workings, the sides of the pillars normally are fractured and fine coal commonly is strewn in the roadways. Hence, a large surface area of coal is available for oxidation and the exothermic reaction produces a rise in temperature, which eventually becomes self-generating.

Most of the factors outlined above exist at this particular mine where spontaneous combustion of coal has occurred since around 1947. For example, surface subsidence has given rise to crownholes and tension cracks, which extend to mine level. These allow free passage of air into the mine workings. In addition, the fracturing and collapse of pillars have provided a greater surface area of coal, and air can permeate these fractures in coal pillars. Some burning pillars have collapsed and weakened the integrity of the remaining pillars, leading to further multiple pillar failure and related subsidence, thus further aggravating the problem. Table 1 indicates that these coals are low-rank bituminous coals. It also shows that the sulphide content and, therefore, pyrite content, frequently is greater than 2%, and so, presumably enhances the process of spontaneous combustion. Although the natural moisture content of the coal prior to the abandonment of the mine is unknown, it is assumed that it is now higher since the mine is flooded or partially flooded in places. Certainly, sulphurous steam emanates from many crownholes almost continuously (Fig. 7) and from some tension cracks. The partially collapsed nature of the workings impedes the rapid flow of air, and so, heat is not readily conducted away from hot spots. Indeed, the existing conditions appear almost ideal for the spontaneous combustion of coal.

Various attempts have been made to control or extinguish the burning coal in the mine. These in-

Table 1
Analysis of coal

(a) From spoil heaps

Coal from spoil heap 1					Coal from spoil heap 2				
Sample no.	Carbon content	Ash content	Sulphur content	Moisture content	Sample no.	Carbon content	Ash content	Sulphur content	Moisture content
1	64.2	22.1	2.25	5.5	1	66.8	22.7	2.35	2.3
2	67.3	21.2	1.81	4.7	2	64.5	21.6	1.79	3.6
3	63.2	24.3	2.31	4.1	3	63.2	29.6	1.85	1.9
4	67.2	25.6	2.10	4.9	4	67.2	26.0	2.17	3.3
5	64.6	27.1	2.21	5.2	5	65.7	27.3	2.28	3.9

(b) From open cast area

Coal from open cast area									
Sample no.	Carbon content	Ash content	Sulphur content	Moisture content	Sample no.	Carbon content	Ash content	Sulphur content	Moisture content
1	61.5	25.4	2.57	4.4	6	61.4	31.2	1.26	4.7
2	66.5	23.9	1.92	3.8	7	58.6	33.0	2.74	3.6
3	62.7	22.0	2.64	4.1	8	65.1	29.6	2.15	2.9
4	64.5	26.9	2.79	4.1	9	66.7	27.9	1.98	3.1
5	62.7	28.5	3.77	3.9	10	68.4	23.8	2.41	4.7

(c) Ash content

Sample no.	SiO_2	Al_2O_3	Fe_2O_3	MgO	CaO	Na_2O	K_2O	TiO_2	P_2O_5
1	52.9	43.3	0.59	–	0.11	0.25	0.76	1.65	0.40
2	40.6	35.7	1.99	0.06	10.2	0.23	0.35	1.75	8.84
3	58.6	38.1	0.22	0.05	–	0.27	0.21	1.93	0.08
4	52.8	43.4	1.48	0.12	0.09	0.12	0.33	1.01	0.21
5	51.6	40.0	2.01	0.07	0.06	0.14	0.39	1.02	0.56
6[a]	52.6	41.4	1.90	0.10	1.26	0.22	0.54	1.11	0.32

[a] Burnt coal.

Fig. 7. Sulphurous stream being emitted from a crownhole.

clude construction of cut-off trenches, which were backfilled with earth, around burning zones (Fig. 3), and injection of water into the workings. In addition, collapsing the workings by subsurface explosions to try inhibiting the access of air has been suggested. As this is just likely to increase the access of air, the suggestion has not been acted upon. None of the efforts have had any real success. Burning appears to have bypassed many of the trenches simply because they were not long enough or wide enough, and water injection has made no impression on the burning coal. The latter is probably because no barriers were placed to stop water that is draining away. In addition, water injection may take considerable time, sometimes years, to be successful. The programme was not continued for a long enough time, a matter of weeks being insufficient. In fact, there can be potential problems associated with water injection. First, carbon monoxide and hydrogen may be produced from water–gas reactions with burning coal, and these would be emitted at the surface. Secondly, the release of heat of condensation causes the temperature to rise in the mine.

5. Spoil heaps

There are two spoil heaps on the site. One covers an area of approximately 56,250 m^2 and has a volume of 843,750 m^3, the respective figures for the other are 66,000 m^2 and 990,000 m^3. Spoil heaps are particularly conspicuous and can be difficult to rehabilitate into the landscape. They are formed of coarse discard, which consists of run-of-mine material and reflects the various rock types that are extracted during mining operations (Bell, 1996). Such discard contains varying amounts of coal that has not been separated by the preparation process. The old spoil heaps on the mine area contain appreciable amounts of coal, enough in fact to consider reworking the dumps (i.e. approximately 20–30%), and are poorly compacted compared with their modern day counterparts. Poor compaction of discard, which is associated with tipping, allows easier permeation of air and water, and so helps the spontaneous combustion process and the development of acid mine drainage.

The chemical composition of some samples of spoil material is given in Table 2. From this, it can be seen that the two principal oxides are silica and alumina. Generally, the content of silica is lower and that of alumina is higher than in spoils in some of the coalfields in western Europe, notably Britain (Anonymous, 1973). Calcium, magnesium, iron, sodium, potassium, and titanium oxides are present in small concentrations. The sulphur content of these spoils averages to approximately 1.5%.

The mineralogical composition of the shale material in the coarse discard, as determined by X-ray diffraction, consisted primarily of kaolinite (up to 76%), quartz (28–29%), and mica (6–21%). Other minerals present include aragonite, dolomite, gypsum, and jarosite. Quartz, mullite, alunite, and tridymite were the principal minerals present in the burnt shale. Minor minerals included rutile and anatase. Pyrite occurs in the shales and coal of the spoil heaps. When pyrite weathers, it gives rise to the formation of sulphuric acid along with ferrous and ferric sulphates and ferric hydroxide, which gives rise to acidic conditions in the weathered spoil. Oxidation of pyrite within spoil heap waste is governed by the access of oxygen, which in turn depends upon the particle size distribution, amount of water saturation, and the degree of compaction. The

Table 2
Analysis spoil heap material

(a) Chemical composition (XRF) of shale (%)

Sample no.	SiO_2	Al_2O_3	Fe_2O_3	MgO	CaO	Na_2O	K_2O	TiO_2	P_2O_5	Sulphur content
1	63.4	32.7	0.96	0.24	0.04	–	1.01	1.43	0.08	0.94
2	80.1	15.7	0.47	0.03	0.03	0.18	0.22	2.64	0.47	1.72
3	61.8	34.0	0.48	0.13	0.02	0.32	0.70	2.19	0.08	1.25
4	65.2	31.0	0.78	0.11	0.07	0.2	0.43	1.59	0.14	2.01
5(B)	52.0	44.6	0.82	0.09	0.05	0.27	0.16	1.93	0.09	0.27

(b) Chemical composition (XRF) of coarse discard

Sample no.	SiO_2	Al_2O_3	Fe_2O_3	MgO	CaO	Na_2O	K_2O	TiO_2	P_2O_5	Sulphur content
1	52.8	44.1	0.38	0.15	0.06	0.29	0.30	1.93	0.07	1.82
2	51.5	44.6	0.79	0.08	0.15	0.07	0.36	2.36	0.14	1.96
3	51.6	44.4	0.36	0.12	0.01	0.15	0.61	2.56	0.13	1.01
4(B)	52.0	44.2	0.58	0.11	0.08	0.19	0.22	2.59	0.09	0.82

(c) Mineralogical composition (XRD) of shale (%)

Sample no.	Quartz	Kaolinite	Illite	Mica	Microcline	Jarosite
1	10	68	0	14	0	6
2	18	53	2	21	2	3
3	14	72	4	6	1	2
4	23	66	1	6	0	3
5	29	58	2	4	2	1

(d) Mineralogical composition (XRD) of burnt shale (%)

Sample no.	Quartz + tridymite	Illite	Microcline	Gypsum	Alunite	Mullite
1	57	0	6	12	9	16
2	78	4	0	0	6	14
3	76	2	3	2	12	10
4	64	0	0	2	8	24
5	69	0	2	0	5	20

B = burnt.

poorly compacted nature of these spoil heaps aids the breakdown of pyrite. The resultant sulphates and sulphuric acid have reacted with clay and carbonate minerals to form secondary products, including aluminium sulphates. Further reactions with these minerals have given rise to tertiary products, such as calcium and magnesium sulphates. Such conditions do not promote the growth of vegetation. Indeed, spoil heaps may contain elements that are toxic to plant life. Some acidic water flows from the spoil heaps and infiltrates the ground. There are no small water courses near the spoils to suffer pollution from such runoff. As the spoil heaps on the mined-out area are not near any urban centre, no attempt has been made at restoration.

Spontaneous combustion of carbonaceous material, aggravated by the oxidation of pyrite, is the most common cause of burning spoil (Beever, 1982). Coal and carbonaceous materials may be oxidized in the presence of air at ordinary temperatures below their ignition point. At relatively low temperatures, an increase in free moisture, which occurs in poorly compacted spoil, increases the rate of spontaneous heating. Oxidation generally takes place very slowly at ambient temperatures, but as the temperature rises, oxidation increases rapidly. As noted above, the lower-rank coal in the spoil heaps is reactive and accordingly susceptible to self-heating. When heated, the oxidation of pyrite and organic sulphur in coal gives rise to the generation of sulphur dioxide. If there is not enough air for complete oxidation, then, hydrogen sulphide is formed. Sulphur gases emanating from these spoil heaps can be readily detected by the nose, especially after rainfall. Other noxious gases are emitted from burning spoil, including carbon monoxide and carbon dioxide.

6. Impact of mining on hydrogeology

The surface subsidence and the associated underground fire have had impacts on both groundwater and surface water hydrology. These impacts include reduced surface runoff, increased groundwater recharge, and deterioration of water quality due to the production of acid mine drainage. Surface runoff is reduced as rainfall collects in collapsed areas after heavy summer rains. The ponded water percolates through subsidence-related tension cracks and crownholes to the underground workings. The workings in no. 2 seam act as aquifers for the percolated water. Water collecting on the western side of the regional anticlinal axis flows to the west and dams up against the boundary pillar with the adjacent mine. However, water moving through the workings on the eastern side of the anticlinal axis flow towards the coal suboutcrop in the vicinity of the Blesbokspruit. Since 1991, after the coal suboutcrop pillar along the eastern boundary of the working was mined, highly acidified groundwater, which previously had been retained behind the pillar, began to seep from the workings. This seepage water flows overland in a series of springs, before merging to enter the nearby stream, that is, the Blesbokspruit. A V-notch flume was installed to measure flow on the stream. Weekly flow readings were recorded for a 1-year period, from December 1994 to November 1995. The total annual volume of water collecting over the mine site catchment area was derived from the rainfall data. The estimated recharge to the coal seam would appear to be around 50% of the volume of rain that falls (Table 3). Fig. 8 suggests that there is a lag time between the heaviest rainfall (January) and maximum flow over the V-notch (August).

Rainwater seeping into the old workings is affected by both the oxidation of pyrite and the presence of the underground fire. As a result, there is a deterioration in water quality. Table 4 includes water quality analyses for samples of water collected at the

Table 3
Rainfall and flow data

Month	Rainfall (mm month^{-1})	Total rainfall in catchment (m^3 month^{-1})	V-notch Reading (cm)	Flow (m^3 month^{-1})
December	140	234,500	6.5	36,119
January	210	351,750	5.2	20,768
February	136	227,800	4.6	15,323
March	158	264,650	6.4	34,087
April	18	30,150	8.6	72,321
May	10	16,750	9.8	99,988
June	30	50,250	9.9	102,538
July	0	0	10.5	118,648
August	0	0	9.3	87,811
September	2	3350	7.4	49,820
October	56	93,800	6.7	38,938
November	30	50,250	7	43,406
Total	790	132,3250	91.8	719,770

Recharge = {Flow / Total rainfall in catchment} × 100 = {719770 (m^3 month^{-1}) / 1323250 (m^3 month^{-1}} × 100 = 54%

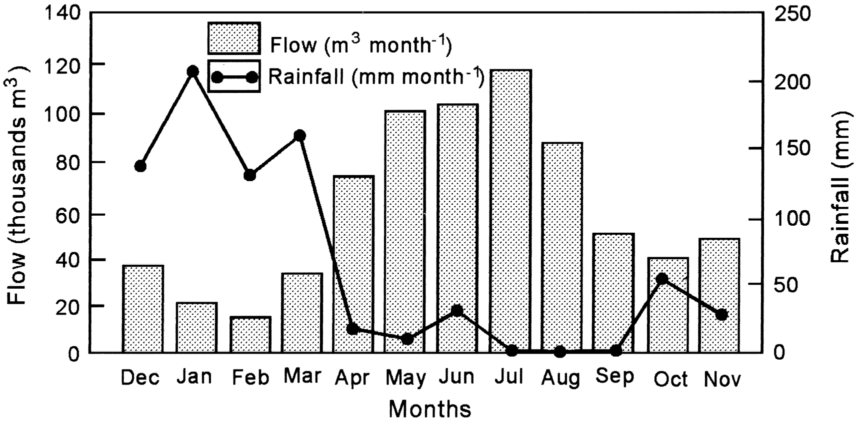

Fig. 8. Monthly rainfall figures compared with seepage flow over the V-notch.

V-notch. Samples were obtained both during the summer (the wet season) and in winter (dry season), and are typical of the quality of water seeping from underground. Analyses of samples taken from drillholes (see Fig. 3) sunk into the no. 2 seam, along with the analyses of water from drillholes sunk at adjacent collieries, are given in Table 5 for comparison.

Inspection of the water quality data at the V-notch and from the mine shows that the waters are highly polluted and that the water in the Blesbokspruit catchment is characterized by low pH and high total dissolved solids. As such, it is capable of mobilizing toxic metal concentrations. The pH values are well below and the total dissolved solids values significantly above the respective crisis limits recommended by the South African Guidelines for Domestic Water (Anonymous, 1993). At such values, there is a danger to health due to dissolved metal ions. Low pH values can be attributed to the formation of sulphuric acid as a product of reactions involving the oxidation of pyrite with the production of acid mine drainage (Bell and Bullock, 1996). In this case, the oxidation of pyrite is enhanced by higher temperatures attributable to burning coal in the old mine. This is supported by the sulphate content which, in all the analyses, is above the crisis limit guidelines, and in some instances is twice that limit. The high sulphate content is not unexpected when compared with the high sulphur content in the coal and associated shale (Tables 1 and 2). Table 4 also shows that the concentrations of aluminium and iron far exceed the crisis limits for drinking water quality in South Africa. Most of the other constituents are near or greater than the maximum permissible limits.

As noted above, there is a time lag between the period of maximum rainfall and maximum runoff. Obviously, this is due to the delay in the contribution made by seepage water from the mine workings. The effect of the time lag between the period of maximum precipitation and maximum flow from the workings is reflected particularly in TDS concentrations. In other words, the salt concentrations for the wettest months, when flow is at its lowest, are generally greater than for the dry months when flow is highest. For example, the relatively low TDS concentration (2968 mg l^{-1}) for August 1996 coincides with the period of maximum flow from the workings. The increased volume of water, therefore, has a dilution effect on the concentration of dissolved salts in the water.

In an attempt to ameloriate the impact of underground water entering the stream, a series of four pollution control ponds was constructed. A sidestream of the seepage water was directed into the upper reservoir while the remaining flow entered the nearby stream below the ponds. Residence time in the ponds is not known but is likely to vary with season. Comparison of water quality data (see Tables 4 and 6) shows that the effect of the cascading part of the seepage water through the decantation ponds is negligible. There is even cause to suspect that aluminium is being leached from soil particles in the ponds as aluminium concentrations leaving the low-

Table 4
Chemical composition of acid mine water from the V-notch and South African guidelines for drinking water quality (Anon, 1993)

Determinand (mg/l)	Sample 1 May 1990	Sample 2 June 1990	Sample 3 July 1990	Sample 4 Dec. 1990	Sample 5 Jan. 1991	Sample 6 March 1991	Sample 7 Dec. 1993	Sample 8 Jan. 1994	Sample 9 Feb. 1994	Sample 10 Dec. 1995	Sample 11 August 1996	Sample 12 August 1996	Sample 13	Recommended) limit (no risk)	Maximum permissible limit (insignificant risk)	Crisis limit (max limit for low risk)
TDS	2749	2575	2575	2843	2760	2082	3376	3038	3575	4844	2968	3202	3604			
EC (mS/m)	293	327	465	421	379	298	424	463	418	471	430	443	340	70	300	400
pH value	2.3	2.4	3.0	1.8	1.9	2.0	2.8	66	2.8	1.9	2.4	2.95	2.8	6–9	5.5–9.5	>4 or <11
Nitrate NO_3 as N	0.05						0.04	0.02	0.18	0.1	0.1	0.1	0.1			
Chloride	91	84	186	120	106	124	179	174	170	310	431	406	611	250	600	1200
Fluoride										0.6	0.5	0.33	0.84	1	1.5	3
Sulphate as SO_4	2361	2239	2697	2462	2330	1692	2722	2378	2897	3250	1610	1730	1440	20	600	1200
Total hardness as CaCO											484	411	377			
Calcium hardness as $CaCO_3$											285	310		20–300	650	1300
Magnesium hardness as $CaCO_3$											199	101				
Calcium	135	115	176	76	132	98	162	179	186	173.8	114.0	124	84	150	200	400
Magnesium	56	50	90	41	55	40	84	90	83	89.4	48.4	49.5	31	70	100	200
Sodium	102	65	200	116	138	114	194	185	200	247.0	326.0	311	399	100	400	800
Potassium									9.4	7.3	9.4	8.9		200	400	800
Iron			140						128	248.3	128	140	193	0.1	1	2
Manganese			18						15	17.9	15	9.9	9.3	0.05	1.0	2.0
Aluminium			86						124		124		84	0.15	0.5	1.0

Table 5
Chemical composition of acid mine water from drillholes sunk at adjacent collieries

Determinand (mg l^{-1})	Middelburg Colliery (2)	Middelburg Colliery (3)	Witbank Colliery (1)	Witbank Colliery (2)	Tavistock Colliery (1)	Tavistock Colliery (2)
TDS	3604	5778	3048	3354	5778	7158
EC (mS m^{-1})	340	355	389	403	355	368
pH value	2.8	2.8	2.65	2.7	2.8	2.9
Nitrate NO_3 as N	0.1	0.19	0.29	0.29	0.21	0.28
Chloride	611	184	951	989	18	4.8
Sulphate	1440	3233	910	1306	3253	3840
Total hardness as CaCO	377	214	106	161	2461	1977
Calcium	84	509	42	40	509	462
Magnesium	31	289	14.9	14.8	289	200
Sodium	399	47	620	755	47	32
Iron	193	198	122	99	198	726
Manganese	9.3	49	5.9	3.9	49	30
Aluminium	84	32	81	87	32	38

est pond are about double than those of water at the V-notch flume.

6.1. Aqueous geochemistry of the Blesbokspruit catchment

The pH value of the waters of the Blesbokspruit varied from pH 2.6, near where the water issuing from the mine, to pH 3.2, downstream of the wetland (Fig. 9). By comparison, two samples taken from the unaffected tributary of the Blesbokspruit, namely, the Prison stream, had values of pH above 7, indicating that they had been unaffected by the acid mine drainage (Table 6). The very low values of pH, which characterized the rest of the Blesbokspruit catchment, indicated a lack of neutralizing capacity and the acidic nature of the waters. The values of electrical conductivity in the catchment area were less than 200 mS m^{-2} for water with a pH less than 3.5 (Fig. 10), whereas the unaffected water of the tributary stream had low electrical conductivity values (Table 6).

High concentrations of aluminium are typical of acid mine drainage water and presumably are derived from alumino–silicate minerals, such as kaolinite and mica, in shales associated with coal seams. The content of aluminium in the Blesbokspruit varied between 80 and 240 mg l^{-1} compared with 0.59–8 mg l^{-1} for stream water that was not polluted. There was a twofold to fourfold decrease in Al concentration in the wetland.

The iron content of the Blesbokspruit was lower than that found in the waters obtained from drillholes, suggesting that much of the dissolved Fe has been precipitated as a result of oxidation and hydrolysis. The equilibrium of iron, according to Karathanasis et al. (1988), is modified in an acid sulphate system. Although goethite and amorphous $Fe(OH)_3$ control Fe levels in most natural aquatic systems, the control of dissolved Fe in acid sulphate-rich solutions appears to be more consistent with the solubility of the iron sulphate mineral jarosite.

Sulphate was the dominant anion in the Blesbokspruit, with concentrations ranging between 128 and 2250 mg l^{-1}. The high concentrations of sulphate are characteristic of water polluted by acid mine drainage. Once the sulphate-rich acid mine drainage entered the Blesbokspruit, the concentration remained fairly constant until it was diluted somewhat by uncontaminated water from the Prison stream.

The chloride ion concentration varied between 80 and 580 mg l^{-1}. Like sulphate, it was derived from the coal and shale in the mine. It would be expected that the sodium and chloride present would have similar origin, but the higher values of chloride suggest that the removal of dissolved sodium occurred as a result of the precipitation of Na–jarosite. In fact, halite was recorded in an X-ray diffractogram of a material precipitated on the surface of soil from acid drainage water (Fig. 11). Calcium and magnesium had similar concentrations in the Bles-

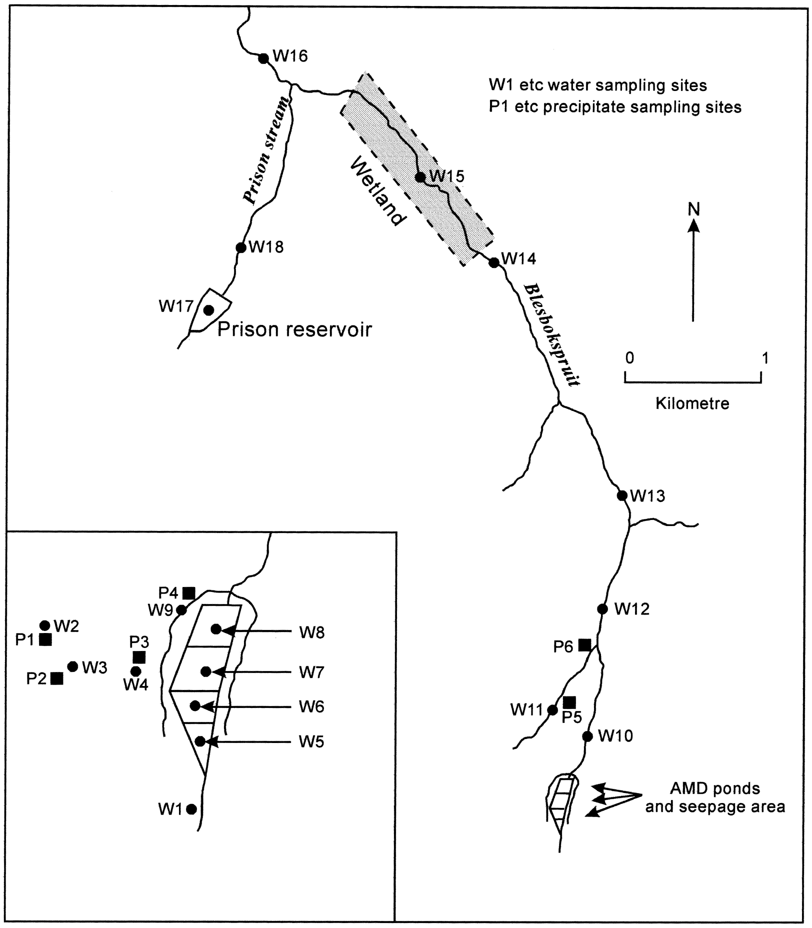

Fig. 9. Location of water and precipitate samples.

bokspruit. The maximum concentration of manganese was 80 mg l^{-1}, and the Blesbokspruit contained significant concentrations of lead and zinc (Table 6). As far as fluoride was concerned, its concentration fluctuated according to the changes of aluminium in the water (Fig. 12). Aluminium–fluoride complexes presumably explain this fluctuation.

The waters of Blesbokspruit contained high concentrations of elements typical of acid mine drainage (Table 6). However, the concentration of elements decreased as the Blesbokspruit flowed through the small wetland. The reduction in the concentration of some elements can be attributed to element retention reactions involving adsorption, precipitation and coprecipitation. Dilution of the water exiting the wetland by uncontaminated water from the Prison stream further decreased the elemental concentrations of the Blesbokspruit. In particular, the precipitation of jarosite and goethite probably accounted for the removal of iron. However, the solubility of aluminium sulphate minerals under acidic conditions would seem to explain the high concentration of aluminium in the water downstream of the wetland.

If natural water is at saturation equilibrium, then the ion activity product should be the same as the solubility product constant. The logarithm of this ratio is referred to as the saturation index. A negative saturation index implies that the waters are unsaturated with respect to a mineral phase, and the mineral should be expected to remain in solution. On the other hand, supersaturated samples have a positive saturation index, the mineral phase being expected to

Table 6
Analyses of Blesbokspruit waters in comparison with that from seepage points and V-notch

Determinand (mg l^{-1})	W1	W2	W3	W4	W5	W6	W7	W8	W9	W10	W11	W12	W13	W14	W15	W16	W17	W18
TDS	578	2280	1748	2097	2356	2625	2295	2341	2504	2164	2526	2215	2225	2232	1070	360	230	180
EC (mS m^{-1})	84	357	371	373	329	390	363	364	378	308	438	316	330	330	166	126	33	23
pH	3.5	2.6	2.7	2.6	2.8	2.7	2.7	2.7	2.6	2.7	2.7	2.8	2.8	2.8	3.1	3.2	7.4	7.3
Chloride	111	468	220	397	486	572	363	380	552	366	425	319	299	271	127	89.6	50.5	48.5
Fluoride	1.1	1.2	1.2	1.2	2.0	1.3	1.3	1.2	1.3	1.3	0.04	1.4	1.5	2.1	1.7	1.2	0.1	0.08
Sulphate	284	1345	987	1226	1416	1512	1509	1511	1470	1417	1587	1487	1512	1510	716	128	45.8	22.9
Alkalinity	nil	nil	nil	nil	nil	nil	nil	nil	nil	nil	nil	nil	nil	nil	nil	nil	73.8	41.1
Calcium	49.2	110	116	117	139	145	136	134	122	111	169	120	151	158	72.8	52.1	23.5	10.6
Magnesium	24.7	43.1	47.1	45.4	56.7	56.9	51.9	51.3	47.0	44.6	64.5	51	64.4	81.4	36.4	25.7	11.6	7.4
Sodium	101	296	360	294	248	302	229	228	298	213	266	228	188	199	111	56.4	23	18.2
Potassium	4.5	11.5	13.4	11.2	4.85	24.5	2.34	33.0	10.5	5.01	8.9	6.7	5.9	7.4	4.3	4.5	3.3	1.9
Iron	1.3	31.6	25.5	46.2	29.6	38.6	34.6	74.4	36.1	27.2	87.1	7.4	19.9	7.0	4.7	1.6	0.61	0.59
Manganese	2.9	10.6	11.5	13.4	12.4	12.4	13.8	12.7	13.4	12	13.7	13.5	19.2	23.1	9.5	6.5	0.09	0.08
Aluminium	7.9	89.4	115	101	96.1	153	190	189	111	132	233	137	161	152	60.3	39.6	0.33	0.27
Silicon	7.8	44.8	44	42.2	31.7	59.9	54.3	55.5	42.2	41.2	64.8	39.4	39.8	36.8	17.1	13.0	0.75	2.67
Copper	1.0	1.4	0.96	1.1	1.0	1.4	1.5	1.1	1.42	1.4	1.5	1.0	1.1	1.0	1.0	0.97	0.94	0.95
Nickel	0.87	1.9	1.8	1.9	1.71	2.2	2.1	2.2	1.9	1.9	2.6	1.8	2.0	1.9	1.3	1.1	BDL	1.1
Lead	1.2	1.6	1.5	1.4	1.4	1.6	1.4	1.4	1.4	1.6	1.6	1.6	1.4	1.7	1.1	1.3	0.96	1.1
Zinc	0.74	2.4	2.7	2.6	1.9	2.3	3.4	2.8	2.6	2.3	4.3	2.2	2.7	2.8	1.4	1.1	0.4	0.4

W2 and W3 samples of seepage point waters.
W4 water sample from V-notch.
W15 Water sample from wetland.
W17 and W18 water samples from uncontaminated Prison stream.
W16 water sample from downstream of confluence with Prison stream.

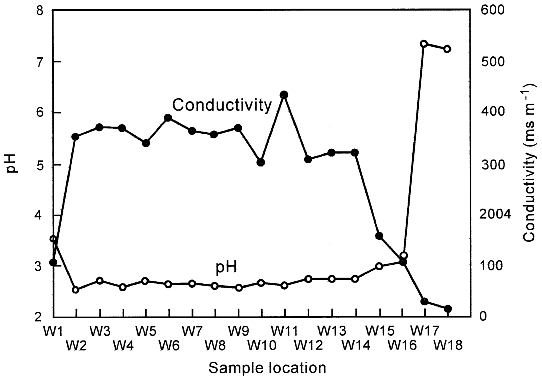

Fig. 10. Relationship of electrical conductivity and pH value of water samples along the Blesbokspruit.

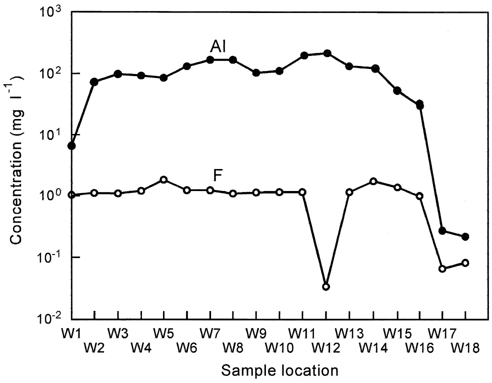

Fig. 12. Changes in the concentration of fluoride compared with aluminium in water samples along the Blesbokspruit.

precipitate out of solution. Saturation indices of around zero indicate that the water is in equilibrium with a particular mineral phase. The MINTEQA2 program was used to determine the saturation indices. Saturation indices of selected minerals were plotted as a function of pH in order to investigate the effects of dissolution or precipitation of minerals on the concentrations of metals in the Blesbokspruit. Water with a low pH is undersaturated with respect to kaolinite and saturated with respect to quartz (Fig. 13A). High concentrations of silica and aluminium correspond with low pH values, where kaolinite is undersaturated. Most water samples with a pH value of less than 3 appear to be supersaturated with respect to jarosite and goethite, and undersaturated with respect to ferrihydrite (Fig. 13B). Gypsum, alunite, and jurbanite may be at equilibrium or slightly undersaturated in water with a low pH (Fig. 13C). Water with a pH exceeding 7 is undersaturated with respect to gypsum and jurbanite, and supersatu-

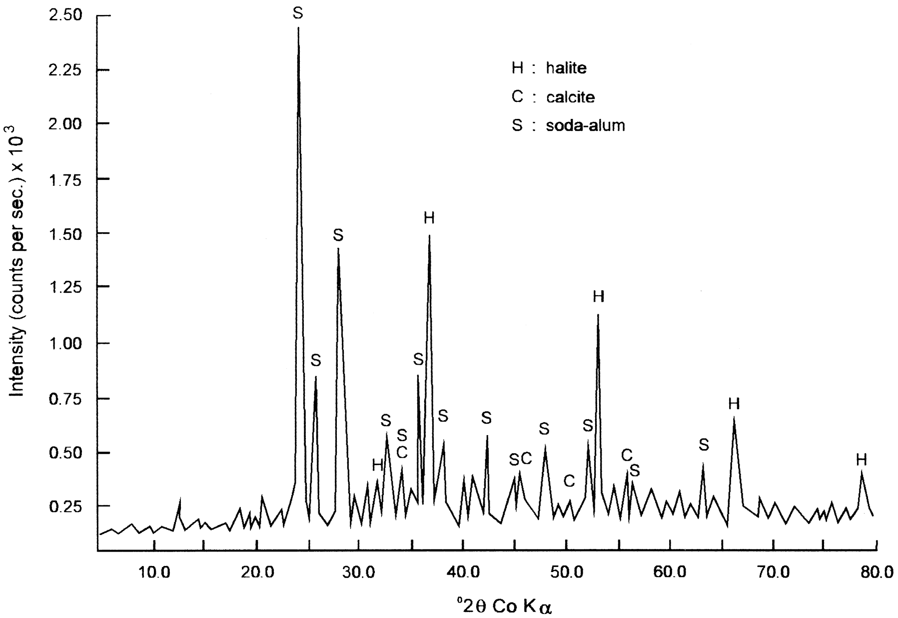

Fig. 11. X-ray diffractogram of salts precipitated from acidified water.

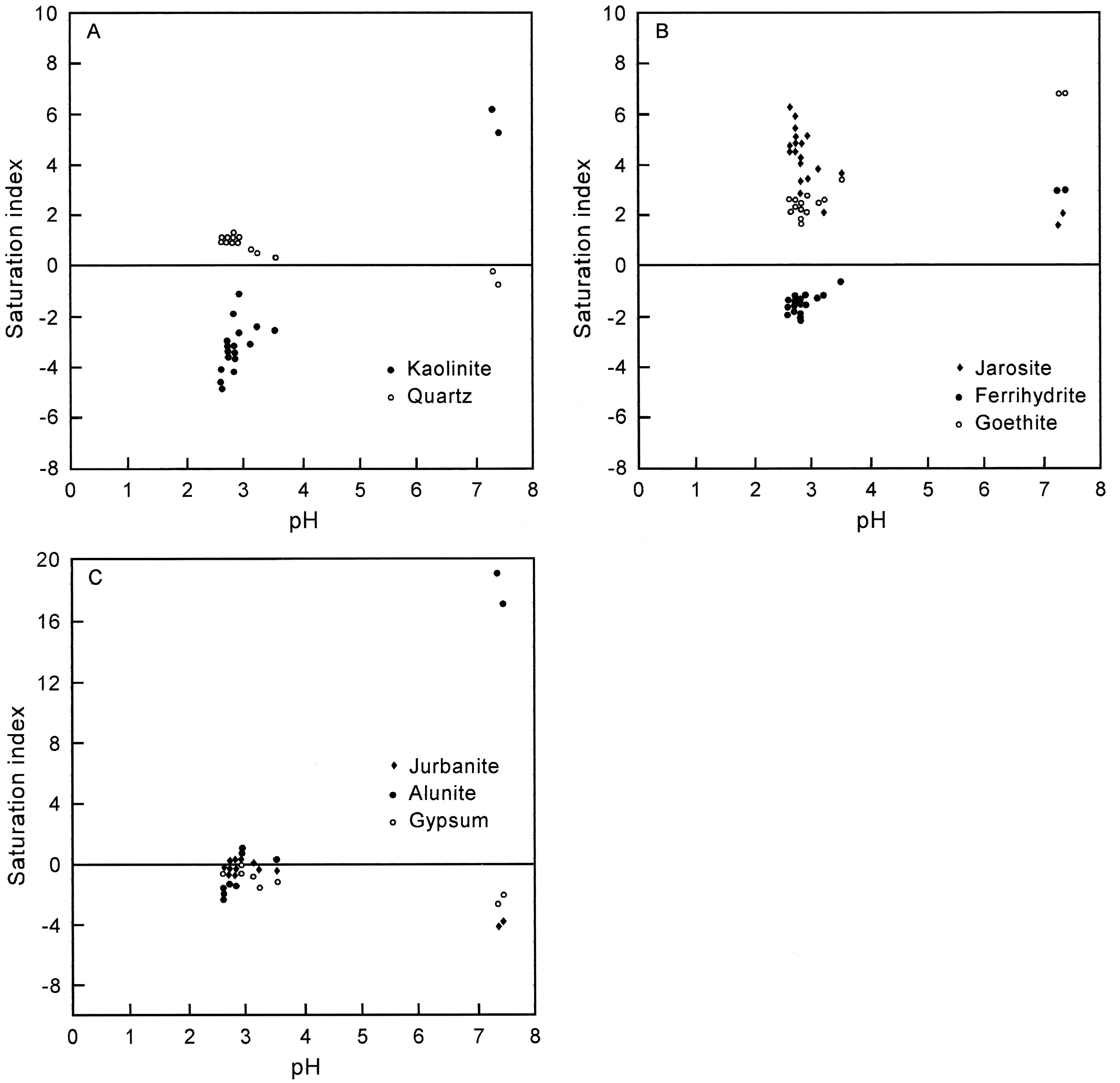

Fig. 13. Saturation indices in relation to pH for (A) kaolinite and quartz (B) ferrihydrite, goethite, and jarosite (C) gypsum, alunite, and jurbanite in the waters of the Blesbokspruit.

rated with respect to kaolinite, jarosite, ferrihydrite, goethite, and alunite. The Blesbokspruit was slightly undersaturated with respect to anglesite ($PbSO_4$), indicating that lead remains in solution in these waters. In fact, the low pH of the water would counteract Pb adsorption to hydrous ferric oxides, such as goethite, and would result in Pb concentrations remaining high (Schwertmann and Taylor, 1989).

The high organic-carbon content in the sediments of the wetland, although partially saturated with aluminium, provides sufficient sorption capacity to attenuate trace metal concentrations. In particular, the concentrations of Zn, Cu, Ni and Mn were reduced (Table 6). Hence, extension of the wetland presumably would further improve the quality of the water exiting from it. Be that as it may, some of additional treatment probably will have to be given

Table 7
Analysis of major elements in precipitates sampled in the Blesbokspruit catchment

In weight percent	Sample no.					
	P1	P2	P3	P4	P5	P6
SiO_2	2.1	35	20	6.5	62	46
TiO_2	0.02	0.42	0.28	0.12	0.33	0.18
Al_2O_3	4.3	7.4	6.7	2.9	3.1	1.6
Fe_2O_3	79	37	65	81	20	40
MgO	0.02	0.09	0.18	0.01	0.12	0.01
CaO	0.06	0.12	0.06	0.02	0.4	0.02
Na_2O	0.12	1.0	0.31	0.25	0.34	0.03
K_2O	0.06	2.01	0.61	0.83	0.59	0.1
BaO	0.003	0.03	0.02	0.01	0.01	0.01
P_2O_5	1.2	0.06	0.16	0.05	0.03	0.03
SO_3	9	11	6.5	8.5	5.3	6.3
Cl	0.22	0.08	0.1	0.1	0.05	0.08

to the water for it to meet acceptable standards, especially in relation to the aluminium content.

6.2. Analysis of precipitate material in the Blesbokspruit catchment

The precipitates in the Blesbokspruit were Fe-rich, and also contained high but variable concentrations of Si, Al, and SO_3 (Table 7). The high Fe content was due to the precipitation of Fe oxyhydroxide, oxide, and sulphate minerals. The SiO_2 content of precipitates was due to the presence of quartz. The low content of CaO relative to SO_3 implied that only trace amounts of gypsum were present in the precipitates. The presence of gypsum was confirmed by SEM-EDS analysis. The X-ray diffractograms of precipitate material taken from near the point of issue of acid mine drainage suggested that ferrihydrite was the dominant mineral precipitating from the iron-rich acidic waters. Ferrihydrite is a widespread and characteristic component of young Fe oxide accumulations, such as ochreous precipitates from ferriferous waters (Schwertmann and Taylor, 1989). In other words, the sudden appearance of Fe^{2+} bearing water at the surface, followed by rapid oxidation and hydrolysis of Fe^{2+}, results in the deposition of ferrihydrite. The presence of an algae mat at the point of issue assisted the oxidation process. Nevertheless, the X-ray diffractograms indicated that jarosite was the dominant mineral in the seepage area, with jarosite being a common mineral in acidic (i.e. when the pH is less than 3.5), high-sulphate environments (Fig. 14). It is possible that various members of the jarosite family of minerals were present (e.g. hydrojarosite, natrojarosite and

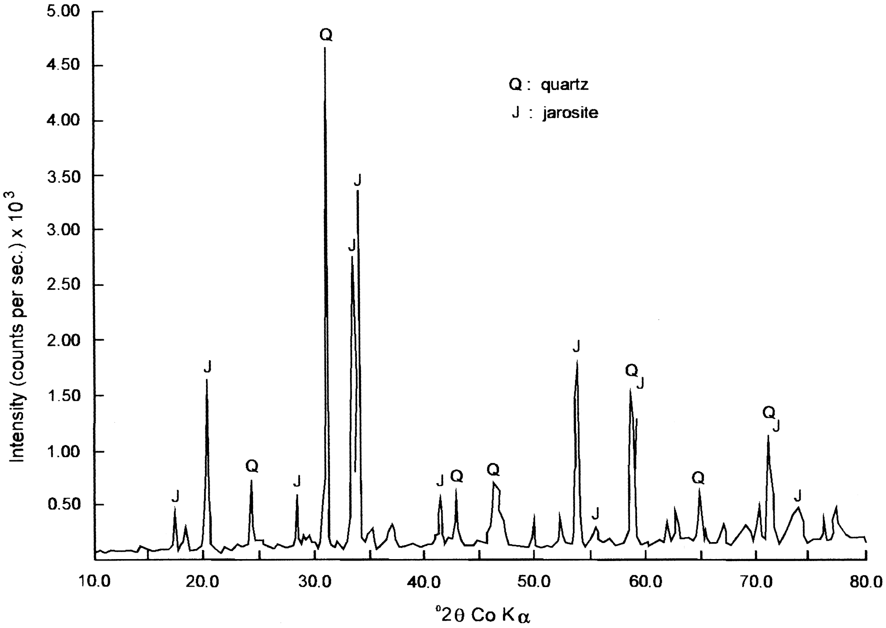

Fig. 14. X-ray diffractogram indicating the presence of jarosite.

plumbojarosite). The precipitation of iron oxide and sulphate minerals by the Blesbokspruit removes and immobilizes dissolved iron.

Some diffractograms indicated the presence of goethite. Goethite, however, is generally a minor constituent of mine drainage precipitates. The precipitation of iron oxyhydroxide and oxides, such as ferrihydrite and goethite from the iron-rich waters seeping out of the mine workings, immobilized dissolved iron as mineral deposits on surfaces of soil, rock or algae in contact with these waters. The formation of small, poorly ordered ferrihydrite crystals generates a large surface area, which makes this phase very active, and compounds are often adsorbed on the surfaces of these particles. The transformation of ferrihydrite to goethite is usually spontaneous because of the lower stability of ferrihydrite (Schwertmann and Fitzpatrick, 1992). This, however, was apparently not the case at the point of issue. The detection of ferrihydrite only in this precipitate suggested resistance to transformation to more stable forms. Carlson and Schwertmann (1981) proposed that such resistance occurs due to the chemisorption of dissolved compounds, including silica and phosphate, to the ferrihydrite surface. These adsorbed compounds not only suppress ordering of the ferrihydrite, but also inhibit the formation of goethite.

Aluminium was present in high concentrations in the acid sulphate waters of the Blesbokspruit. However, no aluminium sulphate or aluminohydroxide minerals were detected in the precipitate samples from the Blesbokspruit catchment, although small amounts of aluminium were detected in SEM-EDS and XRF analyses of the precipitates. In addition, X-ray diffraction analysis suggested the presence of small amounts of soda–alum, $NaAl(SO_4)_2 \cdot 12H_2O$, in the salts (Fig. 11). The suggestion by Nordstrom and Ball (1986) that aluminium behaves conservatively in waters with pH less than 4.6 offers a possible explanation for the virtual absence of aluminium sulphate mineral precipitation in the Blesbokspruit catchment.

The trace element content of the precipitate material is given in Table 8. The accumulation of Zn, Cu, Ni, Mn, Cr, and Pb in the precipitates may be due to the adsorption to iron and sulphate minerals (Schwertmann and Fitzpatrick, 1992). Nevertheless, the adsorption and coprecipitation of trace elements to iron oxide and sulphate minerals did not have a significant effect on the trace element content of the water in the Blesbokspruit. In particular, the low pH of the water counteracted the adsorption of certain dissolved metals, notably lead to iron precipitates. Jarosite acted as an important trace element accumulator. A significant relationship occurred between rubidium and K_2O in the precipitated material. This is due to the ability of rubidium to substitute for potassium in mineral structures. It is, therefore, not surprising to detect high rubidium concentrations in the precipitates containing jarosite. The high lead content of precipitate P4 suggested the possible sorption of Pb to goethite present in this sample. The accumulation of vanadium in precipitates is due to the release of vanadium from coal on oxidation of pyrite, and its subsequent association with algal matter and Fe minerals at the point of issue of acid mine drainage (Edwards et al., 1995).

Table 8
XRF analysis of trace elements in precipitates sampled in the Blesbokspruit catchment

Trace elements ($mg\ kg^{-1}$)	Sample no.					
	P1	P2	P3	P4	P5	P6
Zn	17	44	53	34	18	8.5
Cu	9.4	20	50	39	14	12
Ni	< 0.8	19	11	< 0.8	15	7.1
V	800	62	123	263	198	178
Cr	69	112	570	104	93	81
Mn	37	149	276	34	178	54
Co	6.4	7.3	10	4.0	4.0	2.4
Mo	0.3	0.9	3.3	0.8	2.4	0.9
Rb	3.5	96	24	31	37	5.3
Pb	19	19	24	76	20	4.1
U	1.1	2.4	3.6	4.2	0.7	< 0.9

7. Impact of acid mine drainage on vegetation

Most plants cannot tolerate low pH water because the high concentration of hydrogen ions causes inactivation of enzyme systems, restricting respiration and root uptake of mineral salts and water (Bradshaw et al., 1982). Accordingly, the acidity of the seepage water and the high concentration of total dissolved solids adversely affected vegetation locally. A denuded area of approximately 3 ha exists in the coal suboutcrop area between the eastern boundary of the

Fig. 15. Vegetation decimated in the seepage area by acid mine drainage. Also note the precipitation of salts.

mine and the decantation ponds. In this area, almost all plant life has been killed (Fig. 15). Dissolved aluminium ions, in particular, are regarded as a major cause of plant toxicity in acid soils. As the total aluminium concentration in the seepage water is over 100 mg l^{-1}, aluminium is likely to have played a significant role in the destruction of vegetation in this area. Other than species of algae, no aquatic life appears to exist in the seepage area, in the pollution control ponds or in the stream.

Many species of algae are known to tolerate acid mine drainage, and they appear to play a role in metal attenuation. Intensive algal growth occurs in the seepage area. The green algae belonged to the genus *Mongeotia* and the red-brown algae to the genus *Microspora*. Algae can remove metals from acid mine drainage water, and these algae had accumulated iron, aluminium, copper, nickel, manganese, and lead in their cell walls and cytoplasm. In addition, encrustations of ferrihydrite occurred on dead algae. Hence, these algae, to a certain extent, attenuate the concentration of metals in these waters.

8. Conclusions

Most environmental problems on mine property can be attributed to the mining methods used in the 1930s and 1940s, notably the practise of pillar robbing. Pillar robbing has been responsible for the subsequent collapse of bords and of pillar failure. Subsidence has allowed continuous free entry of air into the workings, and this, together with the heat generated by the oxidation of coal, caused spontaneous combustion of many of the remaining coal pillars. Hence, underground burning has led to further subsidence as pillar strength is weakened. The formation of crownholes and deep tension cracks has increased recharge to the workings. The chemical composition of surface water percolating into the old mine workings is rapidly changed through contact with the pyrite that is oxidized, resulting in the formation of typical low-pH and high-TDS acid mine drainage water. After the removal of a suboutcrop boundary pillar in 1991, acid mine drainage began to seep to the surface. There is a time lag between the summer rainfall and the maximum seepage from the workings. Overland flow of this water towards a local water course, therefore, is strongest in the dry winter months, but the dissolved salt content is diluted. Nonetheless, the acid nature of the water has largely destroyed vegetation in a 3-ha area between the seepage points and the water course. Cascading the seepage water through decantation ponds situated along the water course is ineffective in terms of improving the quality of the water. Red-brown and green algae are the only discernable forms of aquatic life in seepage water springs, in the ponds, and in the stream. A minor contribution to the problem of acid mine drainage is made by the spoil heaps located on the mine site.

The problems outlined in this paper are not unique to this particular mine. Similar problems are found in other abandoned coal mines in this area and elsewhere in South Africa. Often, a responsible ownership cannot be established, and in these cases, responsibility for the mine has been taken over by the Department of Water Affairs and Forestry. In the case of the mine described, the last mining company to operate the site has assumed site responsibility. In fact, water quality is a problem that affects all coal mines in the area. Many local, often ephemeral, streams in this area eventually drain into the Loskop Reservoir and the Olifants River, as does the Bleskokspruit. The Olifants River eventually flows through the Kruger National Park. The catchment area of the Olifants River is thus sensitive, not only from the point of view of tourism and nature conservation, but also because the lower and middle sections of the river are areas of intensive agricultural activity. Water quality management and any attempt at environmental remediation need to be considered within a regional context and will require considerable cooperation between the various mining companies operating in the area and with downstream users. It is likely that remedial measures, or at least holding measures, will be driven by water quality considerations.

South Africa is currently moving away from a uniform effluent standards approach to water quality management towards regulation based on water quality receiving objectives, which take into account the assimilative capacity of the receiving water, as well as the water quality requirements of the downstream user. These principles were embodied in a Water Act recently introduced. All operating mines are now required to submit an Environmental Management Programme Report (EMPR). Sections of this report are legally binding and each mine is required to detail what the residual environmental impacts (including projected impact on water quality) of the mining activities will be and to commit resources to minimize such impacts. It is this more holistic approach to mining, which should mitigate against environmental degradation, such as described in this paper, and ensure that mining activities and environmental protection are not mutually exclusive.

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