CrossMark

Environmental geochemistry of the abandoned Mamut Copper Mine (Sabah) Malaysia

Antony van der Ent · Mansour Edraki

Abstract The Mamut Copper Mine (MCM) located in Sabah (Malaysia) on Borneo Island was the only Cu–Au mine that operated in the country. During its operation (1975–1999), the mine produced 2.47 Mt of concentrate containing approximately 600,000 t of Cu, 45 t of Au and 294 t of Ag, and generated about 250 Mt of overburden and waste rocks and over 150 Mt of tailings, which were deposited at the 397 ha Lohan tailings storage facility, 15.8 km from the mine and 980 m lower in altitude. The MCM site presents challenges for environmental rehabilitation due to the presence of large volumes of sulphidic minerals wastes, the very high rainfall and the large volume of polluted mine pit water. This indicates that rehabilitation and treatment is costly, as for example, exceedingly large quantities of lime are needed for neutralisation of the acidic mine pit discharge. The MCM site has several unusual geochemical features on account of the concomitant occurrence of acid-forming sulphide porphyry rocks and alkaline serpentinite minerals, and unique biological features because of the very high plant diversity in its immediate surroundings. The site hence provides a valuable opportunity for researching natural acid neutralisation processes and mine rehabilitation in tropical areas. Today, the MCM site is surrounded by protected nature reserves (Kinabalu Park, a World Heritage Site, and Bukit Hampuan, a Class I Forest Reserve), and the environmental legacy prevents de-gazetting and inclusion in these protected area in the foreseeable future. This article presents a preliminary geochemical investigation of waste rocks, sediments, secondary precipitates, surface water chemistry and foliar elemental uptake in ferns, and discusses these results in light of their environmental significance for rehabilitation.

Keywords Biodiversity · Floc · Kinabalu · Mamut Copper Mine · Malaysia · Sabah

A. van der Ent (✉) · M. Edraki
Centre for Mined Land Rehabilitation, Sustainable Minerals Institute, The University of Queensland, St Lucia, QLD 4072, Australia
e-mail: a.vanderent@uq.edu.au

A. van der Ent
Laboratoire Sols et Environnement, UMR 1120, Université de Lorraine – INRA, Nancy, France

Introduction

The environmental legacy of abandoned mines depends on geology and climate, as well as past mining and mineral processing practices. Mines operating porphyry Cu–Au systems can potentially pose significant post-mining environmental impacts, because: (i) disseminated ore and large volume of these deposits require extensive open-cut operations with high tonnages of waste rock; (ii) with high throughput rates, large amounts of tailings are consequently produced; (iii) mining wastes contain reactive

Springer

sulphide minerals particularly pyrite and (iv) these ore deposits typically contain igneous host rocks, and their alteration products, with low alkalinity producing capacity (McMillan and Panteleyev 1980; Plumlee et al. 1999; Sinclair 2007; Dold 2014).

The former Mamut Copper Mine (MCM) in Sabah, Malaysia (Fig. 1) was the largest metallic mine in the country at the time of operation. The mine site is an unusual derelict Cu–Au mine on account of its geological setting; a porphyry intrusion wedged between serpentinite rocks (Imai 2000), and its location surrounded by the biodiversity-rich nature reserve Kinabalu Park. The environmental legacy of the MCM site is significant and includes the presence of a large pit lake filled with polluted acid water surrounded by unstable pit walls, as evidenced by extensive cracks in the walls, and large amounts of acid-producing minerals waste. Previous studies on

the rehabilitation and management of minerals waste and site discharge have been undertaken by the local university (Universiti Malaysia Sabah—UMS) and by the state Department for Minerals and Geosciences (JMG) over the last decade. Recently, the Malaysian government (JMG) entered a research partnership with the Korean government (Korea Mine Reclamation Corporation—MIRECO) to test treatment options for the acidic effluent from the MCM site. As part of the ongoing commitment by these organisations, a provisional rehabilitation plan has been developed, but appropriate funding for implementation remains to materialise. In addition, proposals for re-processing the tailings (for Au) have been submitted by business entities to the Malaysian government for approval.

The objective of this study is to provide an overview of the environmental legacy of the MCM site based on a preliminary geochemical investigation of waste rocks,
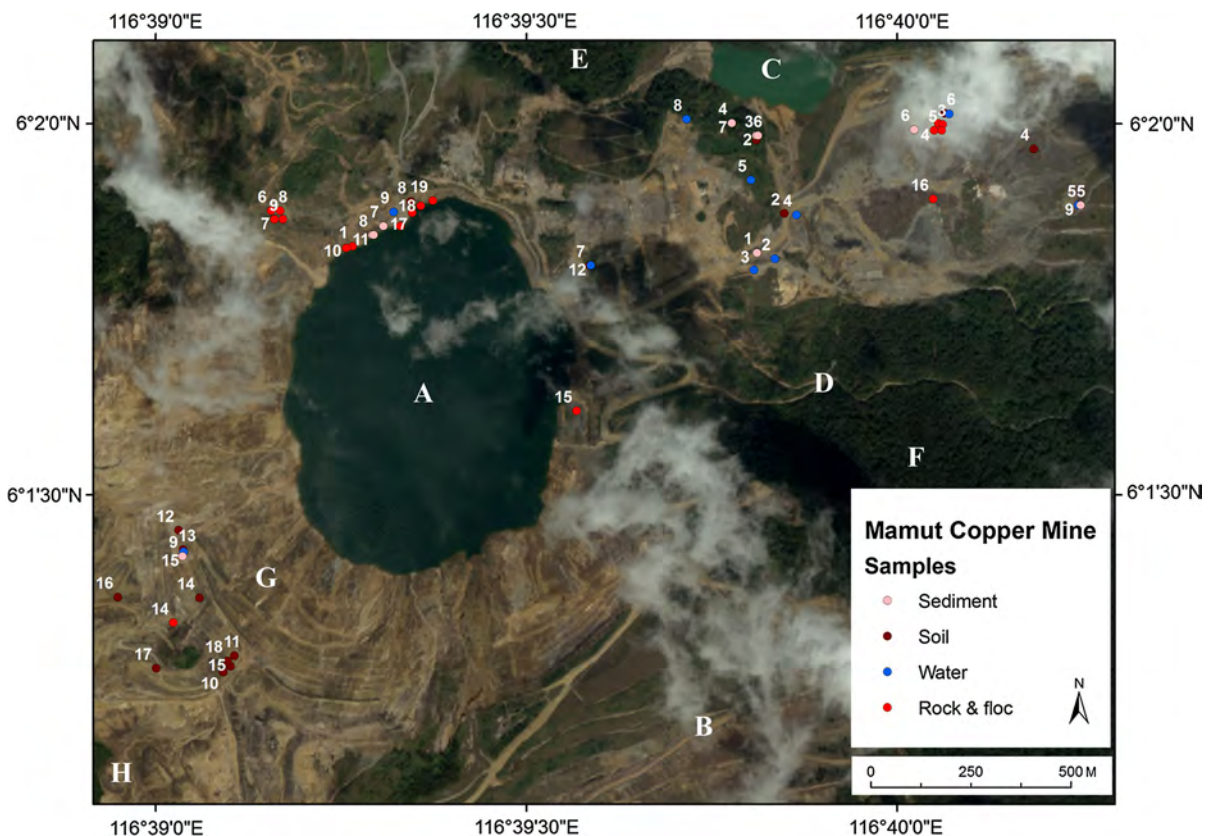

Fig. 1 Location of the MCM site and associated features on Quickbird-2 satellite imagery. *A* Pit lake; *B* overburden waste rock dump; *C* mixing pond pit drainage with Mamut River; *D* main access road to the site; *E* Kinabalu Park; *F* Bukit Hampuan FR; *G* extremely steep slopes (>50 %) with numerous streams entering the pit lake; *H* small farms (abandoned). The sample localities of rock, soil, water and sediment are indicated

soil, sediment, secondary precipitates or "floc", surface water chemistry and foliar elemental uptake in local pteridophytes (ferns). The paper discusses the results of the chemical and mineralogical analyses in light of their environmental significance, and research opportunities with the ultimate aim of site rehabilitation.

Site description

Geography and mining history

The MCM site is located at 06°01′850″N and 116°39′292″E, in Sabah, Malaysia, approximately 68 km from Kota Kinabalu, the state capital of Sabah. The former mine site is located on the south-eastern slope of Mount Kinabalu (4095 m a.s.l.), bordering Kinabalu Park. The site, at an elevation of 1300–1500 m a.s.l., is mainly drained by the Mamut River which discharges into the Lohan River, flowing south-east to east (Keong and Sa 1992). The area has a mean monthly air temperature of 20 °C a daily fluctuation of 7–9 °C throughout the year (Kitayama et al. 1999). Mean annual rainfall measured at Kinabalu Park is 2380 mm (Kitayama et al. 1999), but known to include occasional peaks of up to 700 mm in a week (Keong and Sa 1992). At the tailings storage facility (TSF) near Lohan, the daily temperature ranges from 25 to 35 °C with an annual precipitation of 1500–2500 mm (Keong and Sa 1992).

The mineralised area was first discovered in 1965 by detecting high Cu concentrations in stream sediments of the Mamut and Bambangan Rivers (Newton-Smith 1966; Nakamura et al. 1970). A geochemically anomalous area with >300 $\mu g \ g^{-1}$ Cu in the soil delineated the ore deposit (Woolf et al. 1966; Akiyama 1984). A prospecting licence was subsequently awarded to the Overseas Mineral Resources Development Co. Ltd. of Tokyo, Japan, which undertook exploratory drilling between 1968 and 1970, and a feasibility study in 1971. After the mining concession was granted in 1973, the area was removed from Kinabalu Park for a lease of 30 years (Nakamura et al. 1970). Production started in 1975 by the Overseas Mineral Resources Development Sahah Bhd, which was a joint venture between the Mitsubishi Metal Corporation of Japan and the state government in Sabah (Kosaka and Wakita 1978). In

1987, the company was restructured and renamed Mamut Copper Mining (MCM) Ltd. and in turn acquired by Mega First Corporation (MFC) in 1991, which inherited the mining lease covering an area of 1938 ha until 2003.

From the original ore reserve of 179 Mt @ 0.476 % Cu, 83 Mt @ 0.59 % Cu and 0.5 g t^{-1} Au was mined (Kosaka and Wakita 1978). The mine was operated as an open pit using drilling, blasting and loading with hydraulic shovels and wheeled loaders. The bottom of the mine pit is at the −144 m level (1179 m a.s.l.) with the top at 276 m level (1599 m a.s.l.), and the benches are each 12 m high with an angle of 45° of the pit wall (Akiyama 1984). The MCM site had a mill and flotation plant for processing of the ore. The concentrate was shipped to smelters in Japan from a port 120 km from the MCM site. During its operation, the mine produced an annual average of 120,000 t of concentrate including 28,000 t of Cu, 15 t of Ag and 2 t of Au (Kosaka and Wakita 1978). The mine employed approximately 700 people in 1990 (Keong and Sa 1992). The depth of the ore body (>200 m) combined with steep slopes, relatively weak strength of the rocks and heavy rainfall made mining difficult (Keong and Sa 1992). Between 1975 and 1999, the mine produced 250 Mt of overburden and waste rock, dumped in the upper Lohan Valley, and 150 Mt tailings deposited in the Lohan tailings facility.

Geological setting and ore mineralogy

The Mamut deposit is a porphyry-type Cu–Au deposit genetically associated with a quartz monzonite (or adamellite) porphyry stock, which is one of satellite facies of the K-rich Upper Miocene Mount Kinabalu batholith (Imai 2000). The Mamut porphyry is separated by a north–south fault into an east body and a west body, and both are mineralised (Kosaka and Wakita 1978). Primary sulphide mineralisation includes pyrite, chalcopyrite and pyrrhotite, and with less abundant sphalerite, galena and molybdenite (Akiyama 1984). These minerals occur as disseminations and fracture veining throughout the host rocks, i.e. adamellite porphyry, serpentinite and siltstone (Kosaka and Wakita 1978). Secondary minerals from the oxidation zone include limonite, chalcocite, malachite, azurite, covellite, bornite and cuprite.

The effect of oxidation was confined to 30–40 m from the original surface and is mainly influenced by topography and the water table (Akiyama 1984). The Cu minerals in the ore body are associated with adamellite porphyry (47 %), serpentinite (29 %), siltstone/hornfels (21 %) and granodiorite porphyry (3 %) (Akiyama 1984). The distinctive feature of the Mamut deposit compared to other porphyry Cu deposits in the region was the existence of ultramafic rocks as a wall rock in a large portion of the ore body (Imai 2000). This has environmental implications in the current post-mining landscape, for example the occurence of alkaline and acid drainage leading to the formation of floc.

Tailings storage facility (TSF) at Lohan

The ore was processed, at approximately 550,000 t month^{-1}, on site by communition in the mill and flotation at pH 9–10. The concentrate was piped to the thickener to reduce moisture to 10 % to recover water for reuse in the mill (Azizli et al. 1995). The tailings was piped to the Lohan tailings dam at 350 m a.s.l. located near Lohan Village at a distance of $\sim$15.8 km. The tailings storage facility (TSF) holds 150 Mt of tailings material and covers an area of 397 ha. The dam perimeter is partly constructed of waste rocks and partly from coarse sand separated from the tailings by cycloning (Jopony and Tongkul 2009). Tailings material was fed from the MCM site using gravity via open top steel drop tanks, and the excess wastewater was released into the adjacent Lohan River. As the consequence of severe rain events, flooding of rice paddies with minerals waste has occurred on numerous occasions during the operation of the mine, and the flooding in 1977 destroyed an area of 787 ha, of which 514 ha were planted with rice (Keong and Sa 1992). Discharge from the ore processing plant polluted several rivers, which impacted on the water intake of Ranau, and pollution of the rivers affected fish stocks. The Cu concentrations exceeded water quality standards for years during the operation of the mine (Keong and Sa 1992). Recent investigations concluded that the downstream Mamut River sediments were contaminated with 9–37 µg g^{-1} Co, 41–1348 µg g^{-1} Cu and 15–308 µg g^{-1} Ni, whereas the Cu concentration in the Mamut River sediments has increased from 20- to 38-fold since 2004 (Ali et al. 2015).

Materials and methods

Sample collection and preparation

Samples of waste rock, sediment, floc, soil and surface water were collected from the MCM site in 2012 and 2013. For each collection, GPS coordinates and altitude were recorded. Figure 1 shows an overview of the MCM site, associated features and samples localities on DigitalGlobe QuickBird-2 satellite imagery.

Surface water samples were collected from all major streams and drainages on the MCM site. Each water sample was collected in 50-mL polypropylene tubes after filtering through 0.45-µm syringe filters (Nalgene). The samples were acidified with ultrapure nitric acid (70 %) immediately after collection in the field (ratio: 1 mL:1000 mL). Acidity (pH), electrical conductivity (EC) and total dissolved solids (TDS) were measured in a sub-sample in the field (Hanna Instruments).

All solid samples were gamma irradiated at Steritech Pty. Ltd. in Brisbane following Australian Quarantine Regulations. The analysis of all samples took place at The University of Queensland in Australia.

The soil samples ($\pm$500 g) were packed zipped lock plastic bags, brought to the local field station, air-dried at room temperature (20 °C) to constant weight for 3–4 weeks and sieved to <2 mm using a stainless steel screen to focus on the plant-available soil chemistry.

The rock, sediment and floc samples were also dried at 105 °C for 48 h, individually ground using a Retsch ball-mill with agate jars and balls, and subsequently sieved to <100 µm before analysis.

Foliar samples were collected from pteridophytes ferns from across the MCM site. This group of plants was selected because ferns are numerous as colonisers on minerals waste at the site, and also include a wide range of species. The foliar samples were washed in demineralised water after collection while fresh, over-dried at 70 °C for 72 h and ground in an agate ring mill before digestion as detailed below.

Laboratory analysis

Mineralogy

X-ray diffraction (XRD) spectra were collected with a Bruker D8 Advance X-ray diffractometer with cobalt

target, a diffracted-beam monochromator and scintillation counter detector. The instrumental settings were: 40 kV, 30 mA, 3–80° 2θ, 0.05° step size or increment, with 10 s per step.

Nuclear magnetic resonance (NMR) analysis of floc samples

Selected floc samples were analysed with a Bruker Advance III spectrometer operating at 78.205 MHz for 27Al. The magic-angle spinning (MAS) probe with 4-mm zirconia rotor spinning at 9 kHz. Single-pulse experiment with 1 us pulse with 1000 scans with 3–5 s recycling delay were performed.

Soil elemental chemistry and extractions

Soil moisture, pH and electrical conductivity (EC) were measured in a 1:2.5 soil: water mixture after 2 h agitation on an end-over-end shaker. As a general indicator of mobile metals (e.g. plant-available trace elements), extractable concentrations of metals and metalloids were obtained with Mehlich-3 solution consisting of 0.2 M CH_3COOH, 0.25 M NH_4NO_3, 0.015 M NH_4F, 0.013 M HNO_3, 0.001 M EDTA, at pH 2.50 ± 0.05 according to Mehlich (1984). Samples were agitated in 50-mL tubes for 5 min at 400 rpm and centrifuged for 10 min at 4000 rpm, and the supernatant collected in 10 mL polypropylene tubes. Soil sub-samples (300 mg) were digested using freshly prepared "reverse" Aqua Regia (9 mL 70 % nitric acid and 3 mL 37 % hydrochloric acid per sample) in a digestion microwave (Milestone) for a 1.5-h programme and diluted to 45 mL with ultrapure (MilliPore) water before analysis with ICP-AES. The rock, sediment and floc sub-samples (100 mg) were similarly were digested, but using a mix of 4 mL 70 % nitric acid, 3 mL 37 % hydrochloric acid and 2 mL 32 % hydrofluoric acid per sample in a Milestone digestion microwave using high-pressure closed vessels for 2 h and diluted to 45 mL before analysis with ICP-AES for Al, Ca, Co, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, P, S and Zn.

All soils digests and extraction supernatants were analysed by inductively coupled plasma atomic emission spectroscopy (ICP-AES) (Varian) for Al, Ca, Co, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, P, S and Zn. Each run included sample blanks, sample duplicates and ASPAC (Australasian Soil and Plant Analysis Council) reference soils. The instrument was calibrated using a 6-point multi-element standard prepared in the extraction solution.

Surface water elemental chemistry

The acidified water samples were analysed in the laboratory with inductively coupled plasma mass spectrometry (ICP-MS) for Al, As, Ba, Ca, Cd, Co, Cr, Cu, Fe, K, Mg, Mn, Mo, Na, Ni, P, Pb, S, Sb, Se, Si, Sn, U and Zn. Each run included sample blanks, sample duplicates and internal standards. The instrument was calibrated using a 6-point multi-element standard prepared in the extraction solution.

Foliar elemental analysis

Foliar sub-samples (300 mg) were digested using 7 mL concentrated nitric acid (70 %) and 1 mL hydrogen peroxide (30 %) in a digestion microwave (Milestone) for a 1-h programme and diluted to 30 mL with ultrapure (MilliPore) water before analysis with ICP-AES for Al, Ca, Co, Cr, Cu, Fe, K, Mg, Mn, Na, Ni, P, S and Zn. Titanium was also included in the analysis package as an indicator of potential soil contamination on plant leaf surfaces as Ti concentrations in plants are universally low (<10 μg g^{-1}), but higher in soils (100–500 μg g^{-1}). Each run included sample blanks, sample duplicates and the NIST 1515 (Apple Leaves) reference standard. The instrument was calibrated using a 6-point multi-element standard prepared in the extraction solution.

Statistical analysis

The elemental chemistry data were analysed using the software package STATISTICA version 9.0 (StatSoft) and Excel for Mac version 2011 (Microsoft). The XRD data were analysed with the XPowder software program (version 1.0), and with DIFFRACplus Evaluation Search/Match version 8.0 and the International Centre for Diffraction Data's PDF-4/Minerals database. The NMR results were fitted using PeakFit version 4.11 (SeaSolve Software). The map in Fig. 1 was prepared in ArcGIS version 10 using Quickbird-2 imagery (collected on 29-09-2008, projection in WGS_84, license MapMart #149210).

Results and discussion

Waste rock, sediment and soils

Ultramafic rocks, which are typically rich in ferromagnesian minerals, and their weathering products surround the site, and provide alkalinity. Therefore, acid mine drainage from the leaching of Cu sulphide outcrops and mine wastes are locally neutralised either by direct interaction with these rocks or by mixing with drainages from these rocks. The serpentinite waste rocks are naturally enriched in Cr, Ni and Mn (Table 1), whereas the sulphide waste rocks have high concentrations of chalcophile elements (Cu, Zn) and metalloids (As).

The sediment results for major and trace elements (Table 2) show variable but high concentrations of Cu (0.05–8.5 Wt%). But concentrations of As, Cd, Co, Pb and Se are generally low, for example compared to those of Australian Interim Sediment Quality

Guidelines (ISQG-High; ANZECC 2000). One sample (sample 1) has 6056 μg g^{-1} Zn, but other samples are much lower. Barium concentrations are variable at 1.2–92 μg g^{-1} (Tables 3 and 4).

Ali et al. (2004) showed the level of metals and arsenic in soil samples from the riverbank of Mamut River compared with those of a local river (Kipungit River), which is not affected by mining, were 10–100 times higher. Similarly, for most metals the concentrations were approximately 3–10 times higher in the waters of the Mamut River compared with Kipungit River. More recently, Ali et al. (2011) investigated the quality of stream sediments in Mamut River, downstream from Mamut Mine site. They found that concentration of Cu, Ni, and Pb exceeded both assigned limits of Interim Canadian Sediment Quality Guidelines (ICSQG) and Germany Sediment Quality Guidelines (GSQG) at the sampled locations, and enrichment factor values followed the order Ni > Cu > Co > Zn > Pb. They concluded that

Table 1 Major elements in rock, tailings and floc samples from the MCM site

Sample	Description	As (μg g^{-1})	Co (μg g^{-1})	Cr (μg g^{-1})	Cu (μg g^{-1})	Mn (μg g^{-1})	Mo (μg g^{-1})	Ni (μg g^{-1})	Zn (μg g^{-1})
1	Black precipitate from pit lake	4	14	1389	3020	8060	1.9	683	1072
2	Blue copper floc 1	12	10	707	121000	2936	7.0	440	1787
3	Blue copper floc 2	19	11	493	32160	1224	5.4	847	2574
4	Blue copper floc 3	7	10	1662	10800	2666	2.7	431	765
5	Blue copper floc 4	8	10	56	201000	224	7.1	718	1628
6	Blue copper floc 5	17	16	81	117000	446	9.2	1230	7330
7	Blue copper floc 6	9	11	94	133000	749	4.8	2007	12265
8	Blue copper floc 7	4	7	2100	3640	1738	1.7	623	555
9	Blue copper floc 8	8	14	130	159000	1728	5.1	2115	11825
10	Grey floc from pit lake	12	25	294	8070	276	10.5	33	515
11	Mixed tailings Lohan	6	6	365	1087	742	4.2	120	81
12	Oxidised tailings Lohan	12	6	266	528	423	4.5	60	53
13	Unoxidised tailings Lohan	4	3	373	1137	854	3.4	121	112
14	Waste rock pyrrhotite–pyrite rich	158	0	3	547	15	1.6	40	109
15	Waste rock serpentinite 1	3	7	2293	2	591	1.4	755	49
16	Waste rock serpentinite 2	4	8	2475	7	510	1.1	965	50
17	White floc from pit lake 1	9	11	49	7415	127	8.6	42	440
18	White floc from pit lake 2	13	10	19	14400	93	4.3	32	443
19	White floc from pit lake 3	2	4	18	6671	185	0.5	37	129
20	Yellow floc from pit lake	13	13	1	12300	571	3.6	36	398

Analysis by ICP-AES after high-pressure HF–HNO$_3$–HCl microwave digest

Table 2 Major elements in sediment samples from the MCM site

Sample	Altitude (m a.s.l.)	Al (Wt%)	Ca (Wt%)	Cu (Wt%)	Fe (Wt%)	K (µg g⁻¹)	Mg (Wt%)	Mn (µg g⁻¹)	Na (µg g⁻¹)	Ni (µg g⁻¹)	P (µg g⁻¹)	S (Wt%)	Si (Wt%)
1	1294	2.9	0.1	8.5	0.8	507	0.48	530	485	618	518	0.61	9.76
2	1245	0.9	0.1	0.1	19	3180	1.16	142	682	150	234	2.48	6.22
3	1245	1.4	0.2	0.1	19	5810	0.81	182	1002	161	502	1.57	11
4	1238	2.0	0.1	0.3	2.4	7310	0.18	1179	1910	125	568	0.42	16
5	1198	6.2	0.004	0.04	0.8	228	0.09	63	590	97	167	5.93	9
6	1245	6.5	0.005	5.0	0.2	98	0.05	28	429	93	140	4.83	17
7	1333	0.9	0.3	0.1	3.2	9730	0.63	368	944	266	588	0.52	37
8	1319	7.4	0.02	0.6	1.2	96	0.12	60	458	86	18	2.58	25
9	1458	3.0	0.1	0.0	2.9	11,900	0.19	548	1526	115	422	0.48	43
10	1319	0.2	0.6	0.2	2.9	2885	5.07	6907	523	1217	30	0.18	31
11	1319	0.2	1.0	0.2	3.2	5940	2.71	1230	642	806	71	0.43	34

Analysis by ICP-AES after high-pressure HF–HNO$_3$–HCl microwave digest

Table 3 Trace elements in sediment samples from the MCM site

Sample	Altitude (m a.s.l.)	As (µg g⁻¹)	Ba (µg g⁻¹)	Cd (µg g⁻¹)	Co (µg g⁻¹)	Cr (µg g⁻¹)	Mo (µg g⁻¹)	Na (µg g⁻¹)	Pb (µg g⁻¹)	Sb (µg g⁻¹)	Se (µg g⁻¹)	U (µg g⁻¹)	Zn (µg g⁻¹)
1	1294	1.7	29	1.9	7.0	65	29	485	28	2.0	7.3	1.9	6056
2	1245	3.2	39	1.5	6.4	68	38	682	32	1.5	5.3	30	81
3	1245	3.1	84	3.2	3.5	77	30	1002	12	1.3	2.5	27	84
4	1238	1.7	92	1.3	6.5	56	37	1908	9	0.6	1.2	4.3	153
5	1198	3.8	1.5	1.3	7.8	50	31	590	10	4.6	5.8	1.9	64
6	1245	2.0	1.2	0.8	8.7	50	35	429	4	1.3	4.2	1.3	293
7	1333	1.8	97	1.2	7.4	258	31	944	11	0.4	6.5	3.9	140
8	1319	2.8	1.9	1.0	8.6	16	16	458	18	1.0	4.0	1.8	183
9	1458	2.5	64	0.7	8.8	26	49	1526	10	0.6	8.5	3.3	79
10	1319	1.0	2.0	0.9	8.8	529	12	523	8	5.7	4.7	3.8	1540
11	1319	2.5	1.9	0.6	6.6	829	38	642	17	6.3	1.5	3.7	187

Analysis by ICP-AES after high-pressure HF–HNO$_3$–HCl microwave digest

Table 4 Major and trace elements in soil samples from the MCM site

Sample	Altitude (m a.s.l.)	Al (mg g^{-1})	Ca (µg g^{-1})	Co (µg g^{-1})	Cr (µg g^{-1})	Cu (µg g^{-1})	Fe (mg g^{-1})	K (µg g^{-1})	Mg (mg g^{-1})	Mn (µg g^{-1})	Na (µg g^{-1})	Ni (µg g^{-1})	P (µg g^{-1})	S (mg g^{-1})	Zn (µg g^{-1})
1	1294	11.7	1449	9.0	44	492	45.2	766	13.5	481	112	24	484	3.8	76
2	1282	9.3	964	13	49	387	33.4	1337	11.7	293	149	25	398	3.9	42
3	1242	6.0	535	4	182	1070	58.8	977	11.8	210	49	40	297	3.9	43
4	1228	15.7	5666	24	780	3845	57.1	7271	82.4	1104	54	802	184	16.2	272
5	1198	14.9	7313	24	889	2385	38.4	4973	87.1	515	38	842	44	7.4	82
6	1233	10.3	3636	23	355	2013	38.4	1922	77.2	924	40	445	251	7.4	177
7	1307	5.2	1156	5.8	100	1137	42.7	1094	7.8	166	119	31	423	4.5	59
8	1335	3.6	147	6.9	28	476	50.7	978	4.4	133	45	7.6	421	3.9	26
9	1334	7.6	652	8.0	23	961	49.7	837	8.8	392	37	8.5	493	3.6	107
10	1471	8.0	50	12	22	276	28.1	1135	4.9	175	36	11	154	1.9	26
11	1470	6.2	973	13	17	350	41.8	1566	6.6	275	170	7.2	618	8.5	37
12	1454	9.4	67	15	15	429	26.3	1257	3.8	209	29	17	158	1.1	56
13	1453	8.8	281	8.8	27	369	46.6	1196	6.6	274	37	17	405	2.6	59
14	1465	3.3	4105	11	13	1134	55.8	1201	14.4	3535	36	39	258	6.6	127
15	1431	9.4	308	10	28	372	36.8	944	3.7	176	37	8.0	232	1.6	48
16	1530	3.9	53	13	12	154	26.2	714	2.9	103	37	6.6	238	1.7	18
17	1506	6.1	56	15	18	242	26.9	1308	4.3	200	61	8.8	174	2.0	30
18	1475	4.7	136	28	10	453	67.4	560	4.3	159	25	112	37	182.5	830

Analysis by ICP-AES after HNO$_3$–HCl microwave digest

"the Mamut River sediments were severely contaminated by heavy metals especially Cu, Ni, and Co". The sediment samples of our study were collected from the mines site and were treated differently in terms of particle size and analysis; therefore, results of the two studies cannot be compared directly. Nevertheless, both studies indicate dispersion of contaminated sediments from the mine site. The positive correlation of Mg with Ni ($R^2 = 0.8$) and Cr ($R^2 = 0.6$) shows the common source, which is ferromagnesian minerals in serpentinite rocks. There is no apparent correlation between As, Cu, Pb, and Zn and sulphur concentrations in sediments which shows metals are at least partly adsorbed to the surface of sediments rather than deposited in sulphide or sulphate form.

Soil samples collected from the MCM site were analysed for total elemental concentrations, and for Mehlich-3 extractable concentrations i.e. potentially plant-available (Tables 5 and 6). The soils are generally acid (pH 3.9 ± 0.15). In samples derived from serpentinite rock type (samples 4, 5, 6) total and extractable Ni concentrations are high (696 ± 126 and 100 ± 40 µg g^{-1}, respectively), Soil total Zn concentrations are locally elevated (up to 830 µg g^{-1}), whereas Cu concentrations are generally high but variable (154–3845 µg g^{-1} with a mean of 919 µg g^{-1}). Compared to Australian contaminated land guidelines (NEPM 2013), for example for parks, recreational open space and playing fields, which are often used for closed mines, three samples exceed the guideline value for Cu (2000 µg g^{-1}), one for Mn (3000 µg g^{-1}) and one for Ni (600 µg g^{-1}). A comparison of metals concentrations in Tables 5 and 6 gives the immediate impression that extractable metals form a small portion of total metals. Nevertheless, extractable Al concentrations are at a level (356 ± 35 µg g^{-1}) that might induce phytotoxicity. Extractable concentrations of Co, Cr, Ni and Zn are relatively lower. Extractable Cu concentrations are variable, but high (66 ± 21 µg g^{-1}, up to 349 µg g^{-1}). The acidic range of pH and its correlation with extractable sulphur ($R^2 = 0.6$) would be expected from the sulphide oxidising process and the formation of secondary minerals in the form of sulphates, as also evidenced in the correlation between S and EC as salinity indicator. Plant-available nutrient-concentrations are low (K 16 ± 4.1 µg g^{-1}, P 6.1 ± 1.3 µg g^{-1}).

Elemental analysis of three samples from the tailings storage facility at Lohan (Table 1) showed Si concentrations (24–33 Wt%), confirmed by XRD analysis as Quartz. The base metal concentrations in the tailings material are relatively low with 917 ± 195 µg g^{-1} Cu, 100 ± 20 µg g^{-1} Ni and 82 ± 17 µg g^{-1} Zn. Concentrations of As are also low at 7 ± 2 µg g^{-1}. The unoxidised tailings contain 0.8 % S and the oxidised tailings 0.5 % S. The low pH (7.8) of the tailings material and relatively low S content means that acid-forming potential is probably low. This is confirmed by acid–base accounting (ABA) tests by Jopony and Tongkul (2009) concluding that the tailings has negligible AMD potential. They used the procedures formulated by Skousen et al. (2002), O'Shay et al. (1990) and Sobek et al. (1978).

Floc and other precipitates

The alkaline leachate from overburden heaps mainly of serpentinite rock mixes with acid water draining from the pit lake to form Mg precipitates (Fig. 2h). Elsewhere on the site, Al-floc forms where less alkaline water (pH $\approx$ 6) mixes with the pit lake water (pH 3.4) (Fig. 2f). These flocs are of special interest because they have the capacity to trap other metals such as Cu^{2+}, Pb^{2+} or Zn^{2+} and in suspension can transport these contaminants downstream in rivers (Furrer et al. 2002). For example Table 1 shows high Cu concentrations in floc samples.

Furrer et al. (2002) showed Al_{13} [$AlO_4Al_{12}(OH)_{24}(H_2O)_{12}^{7+}(aq)$] is abundant in Al-rich acidic waters downstream from mine sites and constitute the key molecule in floc formation. The ^{27}AlMAS NMR spectra of white floc samples collected at Mamut mine show peaks near +60 which indicates the presence of $Al(O)_4$ a distinctive feature of Al_{13}. The samples show up to 18.7 % of Al molecules have tetrahedral coordination.

Localised also Cu precipitates occur, mainly in the form of the hydrated Cu sulphate mineral brochantite ($CuSO_4 \cdot 3Cu(OH)_2$). This precipitate contains 9.7 ± 2.6 % Cu and occurs along small drains in overburden spoils (Fig. 2g).

Surface waters

Water samples represent the chemistry of the pit lake, major drainage from the pit, the mixing pond below

Table 5 Mehlich-3 extractable elements and pH and EC in soil samples from the MCM site

Sample	Altitude (m a.s.l.)	pH	EC (µS)	Al (µg g⁻¹)	Ca (µg g⁻¹)	Co (µg g⁻¹)	Cu (µg g⁻¹)	Fe (µg g⁻¹)	K (µg g⁻¹)	Mg (µg g⁻¹)	Mn (µg g⁻¹)	Ni (µg g⁻¹)	P (µg g⁻¹)	S (µg g⁻¹)	Zn (µg g⁻¹)
1	1294	4.2	217	590	212	0.5	60	249	15	194	18	2.7	8.4	253	4.9
2	1282	4.0	477	391	131	0.9	64	258	14	375	11	3.3	12	450	4.7
3	1242	3.9	175	278	48	0.6	55	215	4.3	136	5.4	1.4	3.0	327	1.6
4	1228	5.2	126	144	1124	4.8	349	710	50	764	105	76	3.8	29	70
5	1198	5.2	546	27	1427	9.7	64	490	45	1683	121	176	10	278	24
6	1233	4.6	143	268	551	4.1	243	726	52	991	115	48	4.8	52	45
7	1307	3.8	327	310	323	0.2	102	318	19	121	4.2	1.3	5.7	377	2.4
8	1335	3.4	225	314	27	0.7	22	338	7.4	39	3.2	0.5	1.5	406	0.6
9	1334	3.3	332	466	56	0.4	40	498	4.9	72	14	1.2	2.4	432	6.0
10	1471	3.7	143	525	7.8	0.3	20	220	8.4	37	3	0.7	2.7	277	0.8
11	1470	2.8	952	379	43	0.8	34	1384	1.1	142	23	1.1	24	647	7.2
12	1454	3.9	106	520	18	1.3	24	118	9.0	43	12	1.0	1.8	152	1.3
13	1453	3.4	267	503	28	0.4	30	505	10	71	6.8	1.2	7.4	342	2.0
14	1465	4.2	375	202	510	2.8	14	659	36	1048	334	8.7	10	203	6.6
15	1431	3.3	486	452	170	1.0	16	283	3.5	64	15	1.6	3.3	427	4.1
16	1530	3.7	145	296	4.3	0.0	7.1	247	4.4	20	1.2	0.7	2.6	253	0.3
17	1506	3.6	274	470	21	0.4	34	145	4.0	86	9.0	1.0	3.7	317	1.9
18	1475	3.2	577	277	1.7	0.9	12	345	1.0	18	1.6	0.6	3.1	306	7.2

Analysis by ICP-AES

Table 6 Major elements and pH, EC and TDS in surface water samples from the MCM site

Sample	Altitude (m a.s.l.)	EC (μS)	TDS (mg L^{-1})	pH	Al (mg L^{-1})	Ca (mg L^{-1})	Fe (mg L^{-1})	K (mg L^{-1})	Mg (mg L^{-1})	Mn (mg L^{-1})	Na (mg L^{-1})	P (mg L^{-1})	S (mg L^{-1})	Si (mg L^{-1})
1	1294	1097	542	7.8	0.1	66	0.1	16	131	0.7	1.0	0.03	163	12
2	1273	1896	932	3.2	28	99	16	6.1	178	7.7	2.2	0.4	372	11
3	1264	1875	933	3.2	29	100	11	6.0	180	7.8	2.2	0.03	371	10
4	1284	384	194	5.0	0.2	37	0.2	2.5	19	0.2	1.1	0.1	53	6.7
5	1261	1309	659	7.3	0.1	89	0.0	39	158	0.0	1.2	0.1	182	7.7
6	1245	1896	945	3.1	28	99	6.9	6.0	177	7.5	2.2	0.3	364	9.0
7	1238	168	85	5.0	1.6	10	0.8	0.9	11	0.6	1.8	0.01	26	6.0
8	1235	112	86	6.0	0.5	7.0	0.5	0.8	7.4	0.5	1.9	0.2	14	5.7
9	1202	1721	860	3.2	25	90	6.6	5.6	165	6.8	2.0	0.4	335	8.4
10	1198	1506	726	4.6	18	103	0.5	11	184	5.4	2.0	0.2	328	8.7
11	1273	1607	804	6.9	0.1	150	0.0	25	166	0.0	0.8	0.3	284	5.5
12	1307	2546	1264	5.5	2.0	297	0.0	25	283	5.3	0.6	0.4	613	7.5
13	1334	2027	972	4.1	34	108	0.7	11	269	14	1.7	0.2	480	9.8
14	1319	2002	997	5.1	9.2	174	0.9	14	276	7.0	2.3	0.4	455	7.2
15	1456	1625	812	3.6	74	16	0.6	1.0	175	6.9	0.9	0.1	391	7.8

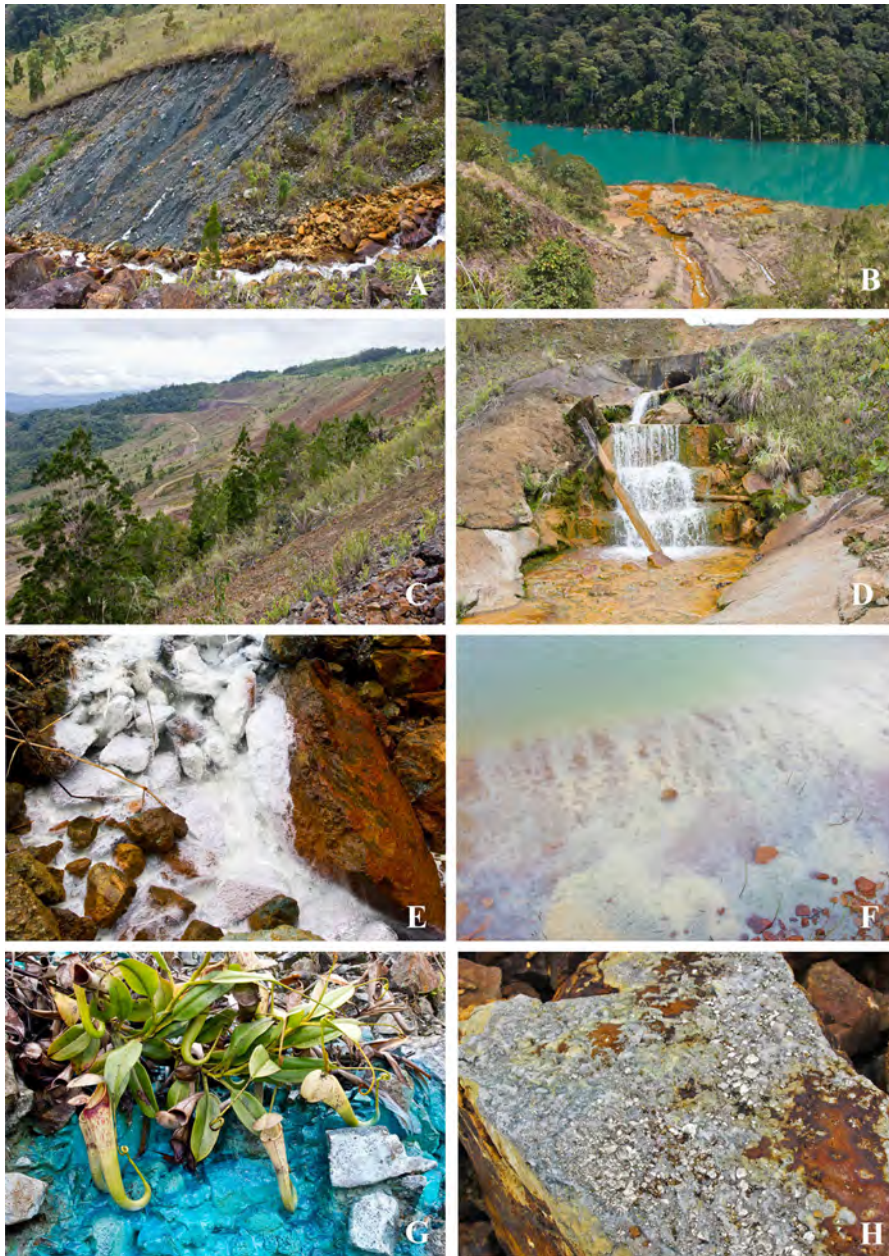Analysis by ICP-MS of acidified samples. pH and EC were measured in the field

Fig. 2 a Serpentinite rock dump; **b** mixing of the pit drainage water with the Mamut River; **c** main waste rock (overburden) dump; **d** drainage from the pit; **e** Mg-floc occurring where alkaline drainage meets acid mine water; **f** Al-floc an associated co-precipitates in the pit; **g** *Nepenthes stenophylla* (Nepenthaceae) growing on Cu precipitates; **h** pyrrhotite-rich waste rock

the pit, undisturbed mountain streams bordering the site and various streams flowing from the waste dumps. The results of ICP-MS analysis results are presented in Tables 5 and 6. The pH values as low as pH 3.1 were measured in acid mine drainage streams running off mineral waste, whereas the pH of undisturbed stream was 5.7. The pH range of surface water on site, from pH 3.1–7.8, reflects the contrasting geochemistry of acid-forming sulphide-bearing rocks and the alkaline serpentinite rocks. The total dissolved

Fig. 3 Formation of the mineral brochantite on boulders overflown with acidic Cu-rich mine water

solids (TDS) are generally high on site (up to 1264 mg L^{-1}) but low in the undisturbed stream (85 mg L^{-1}). Dissolved Al concentrations are high in most samples (up to 74 mg L^{-1}) and negatively correlated with pH ($r = -0.70$). The concentrations of Mn, Cd, Cu, Ni and Pb in surface water samples exceed the Australian guideline values for freshwater aquatic ecosystems (ANZECC 2000). High metal and sulphur (up to 613 mg L^{-1}) concentrations, which are likely in the form of sulphate in the oxidising conditions of the sampling points, are characteristic of AMD streams fed by acidic, and sulphate-rich waters. A comparison of trace metal concentrations from the undisturbed local streams (samples 7 and 8), with those of other samples collected from the MCM site, shows the typical geochemical "fingerprint" of the mine waters from base metal-rich origin (e.g. Seal et al. 2009). For example, Cu concentrations are extremely high in some samples (up to 42 mg L^{-1}). On the other hand, Ni concentrations are also high (up to 1746 µg L^{-1}) reflecting the mixed origin of the stream waters, i.e. the interaction of acid water from sulphide-bearing porphyry rocks and Ni-enriched drainage from serpentinite rocks. Seepages around the mine pit area have low pH (2.9–3.8), high and variable acidity (176–1697 mg $CaCO_3$ L^{-1}), high TDS (302–2673 mg L^{-1}) and high sulphate

(292–2808 mg L^{-1}) (Jopony and Tongkul, 2009). Positive correlations of sulphur concentrations ($r^2 > 0.9$) with EC values indicate the presence of sulphate as the main salt (Figs. 3, 4, and 5).

The pit lake is highly acidic at a mean value of pH 3.4. Previous studies showed that the AMD effluent from the pit lake is diluted to 1:5 by the Mamut River (pH 6.5–6.9), but this dilution only has limited effect on the pH of the water eventually entering the river system, which increases from pH 2.5–3.8 to pH 3.3–4.7 (Isidore and Cleophas 2012). Previous studies also demonstrated that the effects of pollution are significant, with the average concentrations increasing as a result of metal precipitation, for example Cu (105–1606 µg g^{-1}), Zn (157–464 µg g^{-1}) and (64–218 µg g^{-1}) (Ali et al. 2011).

Foliar chemistry of ferns

Pteridophyte samples ($n = 52$ covering 13 species) were collected across the MCM site to serve as a proxy for metal and metalloid biotransfer pathways (Table 7). This taxonomic group of plants was chosen because they are the dominant colonisers on the minerals waste at the MCM site. The terrestrial fern *Pityrogramma calomelanos* (Pteridaceae) is known as an arsenic hyperaccumulator distributed throughout
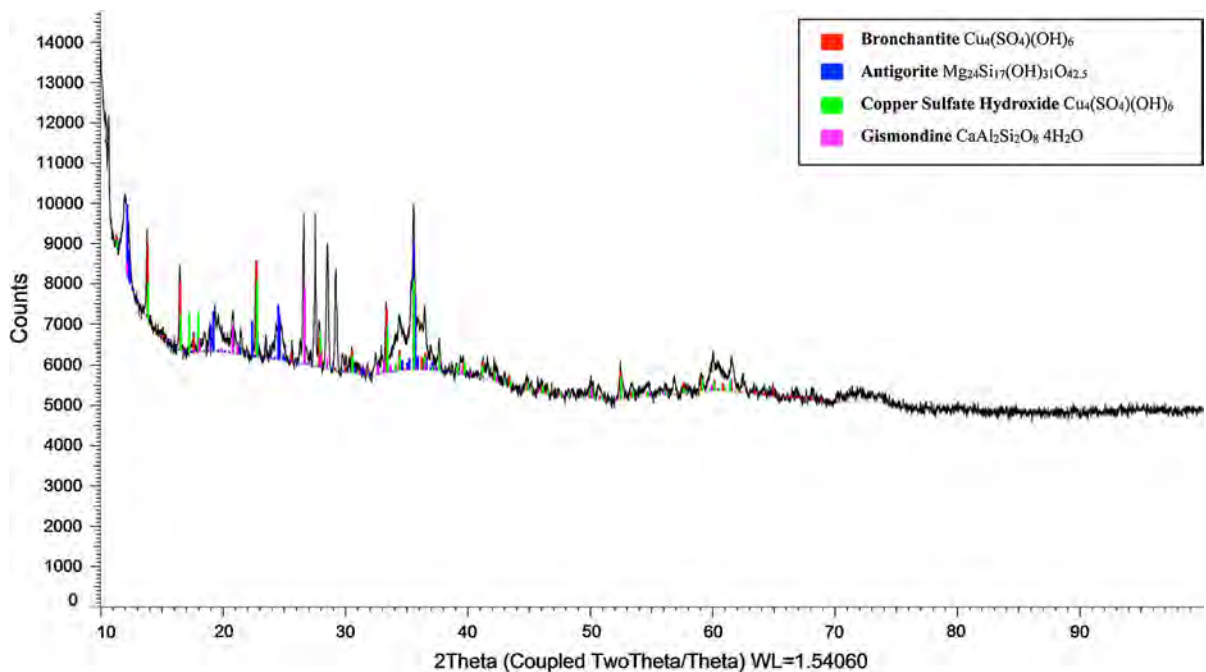
Fig. 4 XRD spectra of selected Cu precipitate sample

SE Asia (Francesconi et al. 2002; Visoottiviseth et al. 2002; Yong et al. 2010). The results here indeed show abnormally high As concentrations in the fronds of this species (39 ± 6.4 μg g^{-1}), much higher than other pteridophytes analysed, but these concentrations do not approach the hyperaccumulator criterion of 1000 μg g^{-1} (Van der Ent et al. 2013). Nevertheless, these results demonstrate that this species has an innate capacity for As accumulation, but soil As concentrations at the MCM site are relatively low and hence not conductive for this species to reach extremely high foliar As concentrations. Al concentrations in two species (*Dicranopteris linearis* and *Matonia pectinata*) are extremely high (at 5018 ± 1248 and 6214 ± 1964 μg g^{-1}, respectively), exceeding the nominal hyperaccumulator criterion at 1000 μg g^{-1} (Van der Ent et al. 2013). Despite the highly variable, but often high soil Cu concentrations, foliar Cu concentrations are within the normal range (11.5 ± 1.5 μg g^{-1}) and have a rather narrow range. Concentrations of Co, Cr, Fe, Mn, Mo, Ni and Zn are all relatively constant and low in all ferns analysed. The terrestrial horsetail *Equisetum ramosissimum* (not a pteridophyte, but also a spore

plant) has unusually high S (at 12466 ± 766 μg g^{-1}) (Table 8).

Plant diversity on the site

The MCM site is surrounded by intact primary tropical montane forest that forms a rich biological reservoir and as a result of advantageous colonisation now hosts a highly unusual flora on the minerals waste, including many rare and threatened plant species. In particular the occurrence of eight species of *Rhododendron* (Ericaceae) and four species of *Nepenthes* (Nepenthaceae), several of which are endemics, is noteworthy.

The highly acidic mineral waste, mainly overburden, on the site is colonised by shrubs such as *Vaccinium retivenium* (Ericaceae), *Macaranga kinabaluensis* (Euphorbiaceae) and *Ceuthostoma terminale* (Casuarinaceae), the latter mainly on less acidic soils. The carnivorous pitcher plant *Nepenthes stenophylla* (Nepenthaceae) is particularly common, together with the fern *Pityrogramma calomelanos* (Pteridaceae), the herb *Dianella ensifolia* (Xanthorrhoeaceae), the orchid *Arundina graminifolia*
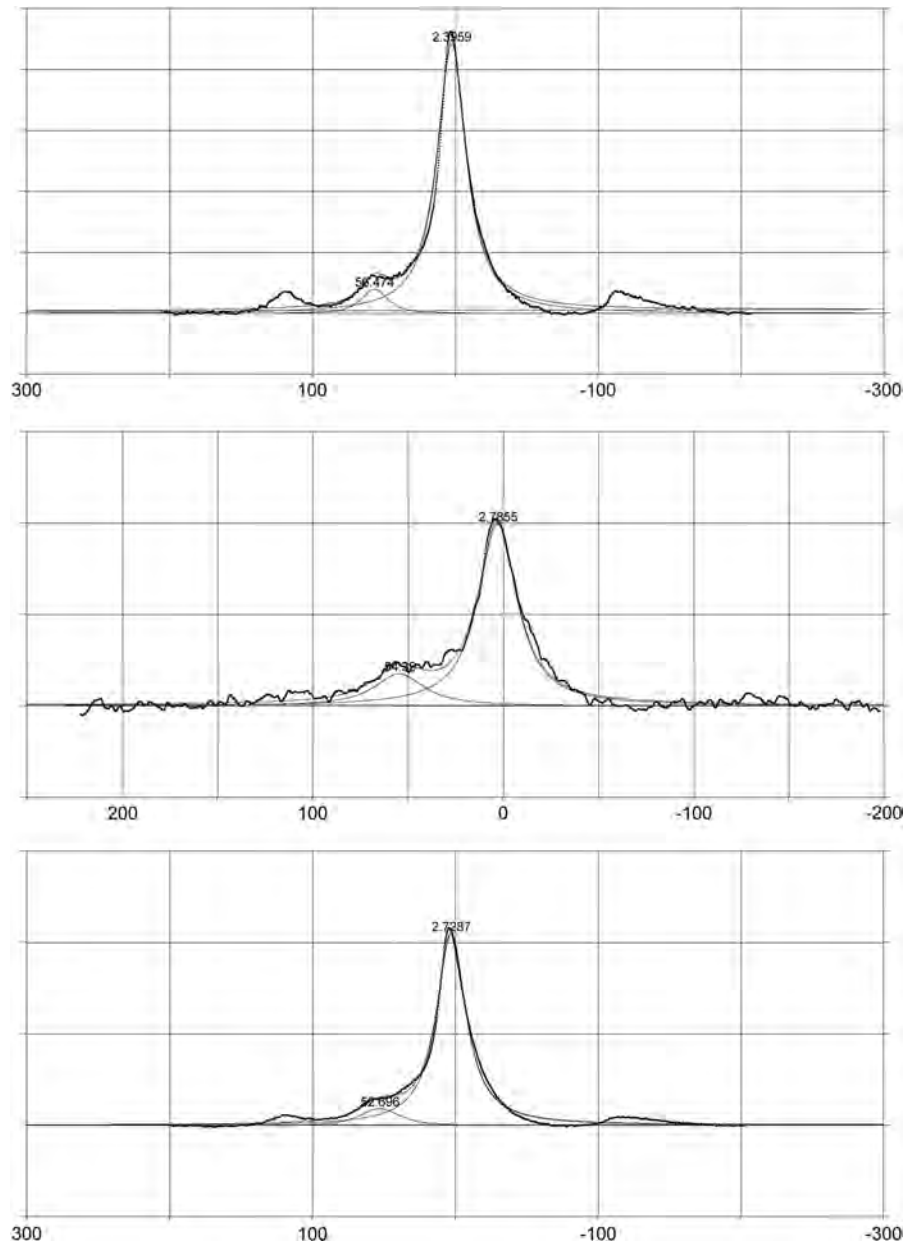
Fig. 5 NMR spectra of selected Al-floc samples

(Orchidaceae), and in wet places *Equisetum ramosissimum* (Equisetaceae) and *Typha angustifolia* (Typhaceae). Introduced *Eucalyptus* spp. (Myrtaceae), which were planted after mine closure, have largely perished; however, *Acacia mangium* (Fabaceae) planted on the tailings is performing well and grown to 10–12 m trees, except at the centre where permanent wet conditions persist.

Rehabilitation

The MCM site presents challenges for environmental rehabilitation due to the presence of large volumes of sulphidic minerals wastes, the very high rainfall and large volume of polluted pit water, requiring comprehensive management actions. The immediate problem is the discharge of poor quality water from the pit and

Table 7 Trace elements in surface water samples from the MCM site

Sample	Altitude (m a.s.l.)	As (µg L⁻¹)	Ba (µg L⁻¹)	Cd (µg L⁻¹)	Co (µg L⁻¹)	Cr (µg L⁻¹)	Cu (mg L⁻¹)	Mo (µg L⁻¹)	Ni (µg L⁻¹)	Pb (µg L⁻¹)	Se (µg L⁻¹)	U (µg L⁻¹)	Zn (mg L⁻¹)
1	1294	5	5	7	24	26	0.8	35	870	16	10	1	0.4
2	1273	8	7	10	31	60	3.6	52	775	14	17	4	1.9
3	1264	8	8	4	33	27	3.7	24	653	21	14	3	1.8
4	1284	9	3	8	24	4	0.4	63	306	23	3	8	0.2
5	1261	0	7	4	24	39	0.4	98	248	18	12	7	0.0
6	1245	5	8	5	33	32	3.6	10	730	21	8	5	1.8
7	1238	6	7	4	23	13	0.3	5	290	27	1	5	0.1
8	1235	7	5	4	24	6	0.1	20	258	7	1	4	0.1
9	1202	11	7	4	31	20	3.3	16	784	14	8	6	1.6
10	1198	25	10	8	30	33	2.7	36	792	31	11	4	1.3
11	1273	6	6	2	23	6	0.2	32	282	7	10	10	0.1
12	1307	2	8	9	28	9	42	32	1746	16	15	14	4.0
13	1334	17	6	5	37	2	8.1	18	863	26	17	8	3.6
14	1319	7	7	3	26	14	3.8	1	550	21	20	6	2.2
15	1456	6	4	12	43	54	14	51	1070	21	13	6	3.1

Analysis by ICP-MS of acidified samples

Table 8 Elemental concentrations of foliar samples of pteridophytes from across the MCM site

Family	Species	Sample (n)	Al (µg g^{-1})	As (µg g^{-1})	Co (µg g^{-1})	Cr (µg g^{-1})	Cu (µg g^{-1})	Fe (µg g^{-1})
Blechnaceae	*Blechnum orientale*	4	148 ± 25	1.0 ± 0.1	4.1 ± 0.7	5.9 ± 1.4	12 ± 1.8	56 ± 12
Polypodiaceae	*Crypsinus soridens*	2	40 ± 5.3	1.2 ± 0.1	1.3 ± 0.1	4.5 ± 1.0	2.9 ± 0.1	18 ± 2.4
Davalliaceae	*Davallia repens*	2	43 ± 17	1.3 ± 0.1	2.2 ± 0.2	5.0 ± 1.8	2.8 ± 0.2	80 ± 52
Gleicheniaceae	*Dicranopteris linearis*	4	5018 ± 1248	1.8 ± 0.2	1.5 ± 0.4	4.8 ± 0.8	7.5 ± 1.0	164 ± 50
Equisetalum	*Equisetum ramosissimum*	2	47 ± 8.8	1.2 ± 0.4	2.7 ± 0.1	6.3 ± 0.5	5.6 ± 0.3	55 ± 3.8
Dennstaedtiaceae	*Histiopteris stipulaceae*	2	96 ± 4.3	1.6 ± 0.1	2.9 ± 0.1	4.4 ± 2.0	8.3 ± 2.3	64 ± 5.7
Matoniaceae	*Matonia pectinata*	4	6214 ± 1964	1.0 ± 0.2	2.2 ± 0.3	5.2 ± 0.7	6.1 ± 1.5	129 ± 83
Nephrolepidaceae	*Nephrolepis cordifolia*	4	100 ± 26	1.4 ± 0.1	3.5 ± 0.9	4.2 ± 0.9	31 ± 13	74 ± 28
Dennstaedtiaceae	*Odontosaria chinensis*	4	531 ± 159	1.7 ± 0.2	2.9 ± 0.3	2.7 ± 1.2	8.0 ± 1.2	96 ± 26
Pteridaceae	*Pityrogramma calomelanos*	17	59 ± 4.6	39 ± 6.4	2.0 ± 0.2	4.5 ± 0.3	16 ± 1.5	40 ± 3.8
Dennstaedtiaceae	*Pteridium esculentum*	2	51 ± 16	0.9 ± 0.1	3.4 ± 1.4	3.4 ± 0.9	5.0 ± 0.6	41 ± 15
Polypodiaceae	*Selliguea triloba*	2	39 ± 6.9	1.4 ± 0.2	2.3 ± 0.2	6.2 ± 0.5	2.5 ± 0.2	19 ± 10
Thelypteridaceae	*Sphaerostephanos lithophyllus*	3	58 ± 4.2	1.6 ± 0.2	2.7 ± 0.4	6.3 ± 0.8	7.1 ± 1.2	64 ± 19

Family	Species	Sample (n)	Mn (µg g^{-1})	Mo (µg g^{-1})	Ni (µg g^{-1})	P (µg g^{-1})	S (µg g^{-1})	Zn (µg g^{-1})
Blechnaceae	*Blechnum orientale*	4	26 ± 2.8	6.6 ± 1.6	59 ± 45	912 ± 254	715 ± 102	17 ± 1.9
Polypodiaceae	*Crypsinus soridens*	2	191 ± 69	5.6 ± 0.5	2.2 ± 1.2	143 ± 21	538 ± 77	9.6 ± 2.7
Davalliaceae	*Davallia repens*	2	314 ± 73	3.8 ± 0.4	8.3 ± 4.4	596 ± 46	1032 ± 356	82 ± 54
Gleicheniaceae	*Dicranopteris linearis*	4	241 ± 86	4.8 ± 1.0	2.6 ± 1.0	273 ± 65	506 ± 84	45 ± 15
Equisetalum	*Equisetum ramosissimum*	2	24 ± 6.0	6.2 ± 0.8	12 ± 4.7	923 ± 175	12466 ± 766	11 ± 1.0
Dennstaedtiaceae	*Histiopteris stipulaceae*	2	849 ± 149	3.8 ± 0.8	3.4 ± 2.7	683 ± 175	914 ± 168	31 ± 2.3
Matoniaceae	*Matonia pectinata*	4	596 ± 172	4.0 ± 0.7	4.7 ± 1.0	433 ± 125	992 ± 184	29 ± 3.9
Nephrolepidaceae	*Nephrolepis cordifolia*	4	26 ± 4.4	4.4 ± 1.1	24 ± 11.2	595 ± 153	1956 ± 771	23 ± 6.9
Dennstaedtiaceae	*Odontosaria chinensis*	4	150 ± 74	4.1 ± 0.6	8.2 ± 2.0	640 ± 67	822 ± 84	19 ± 2.1
Pteridaceae	*Pityrogramma calomelanos*	17	16 ± 1.6	4.2 ± 0.3	4.9 ± 0.7	1796 ± 166	910 ± 51	17 ± 3.4
Dennstaedtiaceae	*Pteridium esculentum*	2	23 ± 0.1	5.8 ± 1.0	5.2 ± 3.2	1364 ± 395	559 ± 23	18 ± 0.7
Polypodiaceae	*Selliguea triloba*	2	255 ± 194	3.2 ± 0.6	5.4 ± 0.5	255 ± 57	3048 ± 2700	12 ± 4.9
Thelypteridaceae	*Sphaerostephanos lithophyllus*	3	28 ± 8.7	4.6 ± 0.5	8.4 ± 1.9	570 ± 37	1051 ± 120	18 ± 4.3

the surface runoff from the site and drainage and seepages from waste rock dumps into the Mamut River, where the mixing of mine water with river water has created a large pool of cloudy greenish water with apparent presence of precipitates in suspension. Leaching of the waste rock dumps and transport of soluble metals is obvious from blue to green Cu precipitates (brochantite) lining the drainages from these dumps.

The original volume of the pit lake can be approximated as 3.2^{E+11} L by calculating the volume of a truncated cone 0.825 km top diameter and 0.6 km bottom diameter ×0.2 km depth = 0.32 km^3 (though note that the current depth of the pit lake is decreasing due to the pit wall caving in). This exceedingly large volume of acid water illustrates the large quality of lime needed for neutralisation treatment. High inflow as a result of high precipitation combined with significant acid-forming rock present means a long-term commitment and costly operation of any treatment scheme. Confounding is that no limestone is locally available, but Jopony and Tongkul (2009) showed locally available materials, namely calcareous sandstone, and calcareous mudstone can potentially be used for treatment of the AMDs at the site. Neutralisation of the AMD with serpentinite rock, locally available, has also been proposed in the past, but even though this rock has high acid neutralising capacity, it also contains significant quantities of Ni, Co, Cr and Mn that would be released upon dissolution. Also, apart from acidity, at 3.7 mg L^{-1} Cu the pit water has a substantial Cu load that needs to be treated.

Previously, during the mine's operation, toxicological studies of local communities using human hair and blood samples for Pb analysis showed no exceeding of safety guidelines (Mokhtar et al. 1994). As this study illustrates, soil and sediment Pb concentrations on the MCM site are relatively low. Ecotoxicological analysis of liver of the toad *Bufo juxtasper* from the Mamut River by Lee and Stuebing (1990) showed highly variable Cu concentrations (4–1020 μg g^{-1}), which were higher than references sites. In addition, toad liver Cd and Ni concentrations from the Mamut River site were also significantly higher compared to reference sites.

Kinabalu Park is renowned for hosting the world's highest plant diversity per unit area with >5000 plant species in <1200 km^2 (Beaman and Beaman 1990; Beaman 2005; Van der Ent et al. 2013). Prior to mining operations (1973), the MCM site formed part of Kinabalu Park but was excised from the Park, and further areas immediately to the south (Bukit Hampuan and Bukit Kulung) were also excised in 1984. As a consequence, these areas were logged and partly cleared for development. Then in 1996, large parts of the area were destroyed by forest fires. In 2006, Bukit Hampuan was re-gazetted as Class 1 Forest Reserve. Therefore, today, the MCM site is mostly surrounded by protected nature reserves.

The environmental legacy of the MCM site, however, prevents de-gazetting and inclusion in either Kinabalu Park or Bukit Hampuan FR in the foreseeable future. Effective mitigation of the negative environmental impacts of the site, and rehabilitation of the site, requires substantial financial commitment because of the large-scale and precipitous morphology of the site and the volume of wastewater to be treated. In brief, the four main priorities for future rehabilitation are: (i) enhance slope instability of the pit walls; (ii) neutralise pit lake discharge entering the Mamut River system; (iii) implement vegetation establishment measures on the minerals waste on the site using local species; and (iv) demolish and remove the remnants of mill buildings and froth flotation installations on the site.

Conclusions

The MCM site has several unusual geochemical features because of the concomitant occurrence of acid-forming sulphide porphyry minerals and alkaline serpentinite minerals, and unique biological features because of the high plant diversity in its immediate surroundings. The geochemical features of the MCM site therefore provide unique opportunities for understanding the post-closure acid mine drainage neutralisation processes, and particularly the role of mafic silicates such as chlorite in the remediation of acid mine drainage and the natural attenuation of heavy metals and arsenic. On the other hand, the naturally occurring rapid colonisation and establishment of plant species on minerals waste at MCM, including aspects of metal tolerance, provide excellent opportunities for further research to better understand metallophytes in the context of mine closure and rehabilitation.

References

Akiyama, Y. (1984). A case history-exploration, evaluation and development of the Mamut porphyry Cu deposit. *Geological Society Malaysia Bulletin, 17*, 237–255.

Ali, B.N.M., Abdullah, M.H., & Yik, L.C. (2011). Application of geoaccumulation index and enrichment factor for assessing metal contamination in the sediments of Mamut River, Sabah. In *National geoscience conference, 11–12 June 2011*. Johor: The Puteri Pacific Johor Bahru.

Ali, M. F., Heng, L. Y., Ratnam, W., Nais, J., & Ripin, R. (2004). Metal distribution and contamination of the Mamut River, Malaysia, caused by Cu mine discharge. *Bulletin of Environmental Contamination and Toxicology, 73*, 535–542.

Ali, B. N. M., Lin, C. Y., Cleophas, F., Abdullah, M. H., & Musta, B. (2015). Assessment of heavy metals contamination in Mamut river sediments using sediment quality guidelines and geochemical indices. *Environmental Monitoring and Assessment, 187*, 4190.

Australia and New Zealand Environment Conservation Council (ANZECC). (2000). *Australian water quality guidelines for marine and freshwaters*. Canberra: Australian Government.

Azizli, K. M., Yau, T. C., & Birrel, J. (1995). Design of the Lohan Tailings Dam, Mamut Copper Mining Sdn. Bhd., Malaysia. *Minerals Engineering*, *8*, 705–712. doi:10.1016/0892-6875(95)00031-k.

Beaman, J. H. (2005). Mount Kinabalu: hotspot of plant diversity in Borneo. *Biologiske Skrifter, 55*, 103–127.

Beaman, J. H., & Beaman, R. S. (1990). Diversity and distribution patterns in the flora of Mount Kinabalu. In P. Baas, K. Kalkman, & R. Geesink (Eds.), *The plant diversity of Malesia* (pp. 147–160). Dordrecht: Kluwer Academic Publishers.

Dold, B. (2014). Evolution of Acid Mine Drainage formation in sulphidic mine tailings. *Minerals, 4*, 621–641.

Francesconi, K., Visoottiviseth, P., Sridokchan, W., & Goessler, W. (2002). Arsenic species in an arsenic hyperaccumulating fern, *Pityrogramma calomelanos*: A potential phytoremediator of arsenic-contaminated soils. *Science of the Total Environment, 284*, 27–35.

Furrer, G., Phillips, B. L., Ulrich, K.-U., Pöthig, R., & Casey, W. H. (2002). The origin of aluminum flocs in polluted streams. *Science, 297*, 2245–2247.

Imai, A. (2000). Genesis of the Mamut porphyry Cu deposit, Sabah, East Malaysia. *Resource Geology, 50*, 1–23.

Isidore, F., Cleophas, F., Bidin K., & Abdullah M.H. (2012). Acid mine drainage dilution and heavy metal removal in temporary settling pond of Mamut Ex-Cumine, Ranau. In *UMT 11th International Annual Symposium on Sustainability Science and Management 09th–11th July 2012, Terengganu.*

Jopony, M., & Tongkul, F. (2009). Acid mine drainages at mamut Cu mine, Sabah, Malaysia. *Borneo Science, 24*, 83–94.

Keong, Y. P., & Sa, T. T. (1992). Land use and the environment in the South Kinabalu Highlands, Malaysia. *Malaysian Journal of Tropical Geography, 23*, 103–118.

Kitayama, K., et al. (1999). Climate profile of Mount Kinabalu during late 1995 - early 1998 with special reference to the 1998 drought. *Sabah Parks Nature Journal, 2*, 85–100.

Kosaka, H., & Wakita, K. (1978). Some geologic features of the Mamut porphyry Cu deposit, Sabah, Malaysia. *Economic Geology, 73*, 618–627.

Lee, Y. H., & Stuebing, R. B. (1990). Heavy metal contamination in the River Toad, *Bufo juxtasper* (Inger), near a Cu mine in East Malaysia. *Bulletin of Environmental Contamination and Toxicology, 45*, 272–279.

McMillan, W. J., & Panteleyev, A. (1980). Ore deposit models—1. Porphyry Cu deposits. *Geoscience Canada, 7*, 52–63.

Mehlich, A. (1984). Mehlich-3 soil test extractant: A modification of Mehlich-2 extractant. *Communications in Soil Science and Plant Analysis*, *15*(12), 1409–1416.

Mokhtar, M. B., Awaluddin, A. B., Fong, C. W., & Woojdy, W. M. (1994). Lead in blood and hair of population near an operational and a proposed area for copper mining, Malaysia. *Bulletin of Environmental Contamination and Toxicology*, *52*, 149–154.

Nakamura, T., Miyake, T., Kanao, N., & Tomizawa, N. (1970). Exploration and prospecting in Mamut mine, Sabah, Malaysia. *Mining Geology, 20*, 100.

National Environment Protection Measures (NEPM). (2013). Accessed June 08, 2015, (http://www.ephc.gov.au/nepms).

Newton-Smith, J. (1966). Geology and copper mineralisation in the Mamut River area, Kinabalu. *Borneo Region, Malaysia Geological Survey Annual Report for 1965, 1966*, 88–96.

O'Shay, T. A., Hossner, L. R., & Dixon, J. B. (1990). A modified hydrogen peroxide oxidation method for determination of potential acidity in pyritic overburden. *Journal of Environmental Quality, 19*, 778–782.

Plumlee, G. S., Smith, K. S., Montour, M. R., Ficklin, W. H., & Mosier, E. L. (1999). Geologic controls on the composition of natural waters and mine waters draining diverse mineral-deposit types, Chapter 19. In L. H. Filipek & G. S. Plumlee (Eds.), *The environmental geochemistry of mineral deposits, Part B: Case studies and research topics, reviews in economic geology* (Vol. 6B, pp. 373–432). Littleton, CO: Society of Economic Geologists, Inc.

Seal II, R.R., Piatak, N.M., Levitan, D.M., Hageman, P.L., & Hammarstrom, J.M. (2009). Comparison of geochemical characteristics of modern-style mine waste from a variety of mineral deposit types for insights into environmental challenges associated with future mining. In *Proceedings of Securing the Future and 8th ICARD, 23–26 June 2009* (pp. 1–10), Skellefteå.

Sinclair, W.D. (2007). Porphyry deposits. In W.D. Goodfellow (Ed.), *Mineral deposits of Canada: A synthesis of major deposit-types, District Metallogeny, the evolution of geological Provinces, and exploration methods: Geological association of Canada* (vol. 5, pp. 223–243). Mineral Deposits Division, Special Publication.

Skousen, J., Simmons, J., & Ziemkiewicz, P. (2002). The use of acid-base accounting to predict post-mining drainage quality on West Virginia surface mines. *Journal of Environmental Quality, 31*, 2034–2044.

Sobek, A., Schuller, W., Freeman, J.R., & Smith, R.M. (1978). Field and laboratory methods applicable to overburden and minesoils. In US Environmental Protection Agency. Cincinnati, OH: EPA-600/2-78-054.

Van der Ent, A., Baker, A. J. M., Reeves, R. D., Pollard, A. J., & Schat, H. (2013). Hyperaccumulators of metal and metalloid trace elements: Facts and fiction. *Plant and Soil, 362*, 319–334.

Visoottiviseth, P., Francesconi, K., & Sridokchan, W. (2002). The potential of Thai indigenous plant species for the phytoremediation of arsenic contaminated land. *Environmental Pollution, 118*, 453–461.

Woolf, D.L., Tooms, J.S., & Kirk, H.J.C. (1966). Geochemical survey in the Labuk Valley, Sabah. *Borneo Region, Malaysia Geological Survey Annual Report* (pp. 212–226).

Yong, J. W., Tan, S. N., Ng, Y. F., Low, K. K., Peh, S. F., Chua, J. C., et al. (2010). Arsenic hyperaccumulation by *Pteris vittata* and *Pityrogramma calomelanos*: A comparative study of uptake efficiency in arsenic-treated soils and waters. *Water Science and Technology, 61*, 3041–3049.