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PROCEEDINGS OF THE THIRD BRUKUNGA WORKSHOP

Quantifying the degree of ecological detriment in the Dawesley Creek/Bremer River system

Prepared within the Managing Mine Wastes Project

February 2000

Ray Cox. Mining Operat.

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Australian Nuclear Science and Technology Organisation

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Quantifying the degree of ecological detriment in the Dawesley Creek/Bremer River system

Held at the Australian Water Quality Centre (SA Water), Bolivar, South Australia, 24th November 1999

Edited and prepared by

Scott J. Markich

Prepared within the Managing Mine Wastes Project

A project in the Core Business Area of Competitiveness and Ecological Sustainability of Industry

Project Leader: John W. Bennett

Cover photo: Bremer River, downstream of the confluence with Mount Barker Creek (courtesy of John M. Ferris, ANSTO)

February 2000

CONTENTS

PREFACE

These proceedings summarise the outcomes of the Third Brukunga Workshop on the impacts of acid rock drainage (ARD) from the Brukunga mine site on downstream aquatic ecosystems, particularly the Dawesley Creek/Bremer River system, located about 50 km west of Adelaide in the Mount Lofty Ranges, South Australia.

The open-cut mining operation at Brukunga, which commenced in 1955 under the ownership of Nairne Pyrites Pty Ltd, saw the conversion of pyrite (iron sulfide) to sulfuric acid for the manufacture of superphosphate fertiliser. Mining operations ceased in 1972 when sulfur production became uneconomical. In 1977 the South Australian government accepted responsibility for the Brukunga mine site and rehabilitation of impacted areas, following an agreement with Nairne Pyrites Pty Ltd.

Surface water in Dawesley Creek is a valuable resource for agricultural (*e.g.* irrigation for crops, including commercial vineyards, and water for livestock) and recreational (*e.g.* swimming) activities downstream of the mine site. However, these activities have been virtually precluded by the poor water quality, let alone the sustainability of the aquatic ecosystems.

Brukunga has been selected as a site to study the ecological effects of ARD, as part of the Managing Mine Wastes Project (MMWP) at ANSTO. The MMWP is part of ANSTO's core business area of Competitiveness and Ecological sustainability of Industry. The work at Brukunga complements ANSTO's ecological studies at Rum Jungle (NT). Due to the close proximity of the Brukunga mine site to Adelaide, and to make the best use of available resources, ANSTO has been actively collaborating with stakeholders (local universities, state government departments, industry and local community groups) to meet the following objective: *To quantify the degree of ecological detriment in the Dawesley Creek/Bremer River system.*

The information obtained over the past three years contributed to the broader objectives of:

- defining an acceptable degree of ecological detriment for various regions of the Dawesley Creek/Bremer River system;
- determining what level of improvement in water quality (*e.g.* pollutants loads and/or concentrations) is required, and
- establishing what engineering controls can be used to achieve the required level of improvement.

This workshop provided a venue to present research results of the previous twelve months, but more importantly, to summarise research findings over the past three years and for stakeholders to discuss the direction of future work.

Scott Markich

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WORKSHOP PROGRAM

TIME	SPEAKER	TITLE
09:00	Scott MARKICH (ANSTO)	Welcome and introduction
09:15	Hume MacDONALD (Brukunga Mine Site Remediation Board)	Brukunga Mine Site Remediation Board action plan for overcoming pollution from the mine
09:45	Ivan DAINIS (Department of Premier and Cabinet)	A conceptual model of the chemical loads in Dawesley Creek arising from acid sulfates in the Brukunga quarry
10:15	Graham TAYLOR (CSIRO)	Acid mine drainage at Brukunga: Impacts and remediation
10:45	MORNING TEA	
11:00	Karyn WILDE (ANSTO)	Measurement of impact of acid rock drainage on microbial communities using community level physiological profiling
11:30	Peter HOLDEN (ANSTO)	Phospholipid fatty acid analysis as a measure of impact of acid rock drainage on microbial communities in sediment: Comparison with other measures
12:00	Rebecca EDWARDS (Flinders University – ANSTO scholar)	Fungi at Brukunga: Community structure and metal tolerance
12:30	Adam SINCOCK (University of Adelaide)	Benthic diatoms as indicators of water quality in Dawesley-Bremer catchment: A system affected by acid mine drainage, sewage pollution and salt–Phase II
13:00	LUNCH – AWCC Cafeteria	
14:00	John FERRIS (ANSTO)	Algae at Brukunga: What do we know?
14:30	Peter SCHULTZ (AWQC, SA Water)	Brukunga acid mine drainage: The macroinvertebrate story
15:00	John TWINING (ANSTO)	Application of AQUARISK to the Dawesley Creek system using water quality and macroinvertebrate monitoring data: Implications for ecosystem rehabilitation strategy
15.30	AFTERNOON TEA	
15:50	Ian RITCHIE (ANSTO)	Current knowledge base for decision making at the Brukunga mine site
16:50	Meeting close	

Hume MacDonald – Brukunga Mine Site Remediation Board

Brukunga Mine Site Remediation Board action plan for overcoming pollution from the mine

Vision
Water quality in Dawesley Creek emanating from the Brukunga Mine site is suitable for domestic (excluding human consumption), stock and primary production purposes.

Key objectives

Objective 1	Identification and documentation of the extent of problem
Objective 2	Community consultation
Objective 3	Solutions to fix the problem identified and implemented
Objective 4	The extent of the problem and its funding requirements sold to the Minister and Government.

Brukunga mine site remediation board terms of reference
The role of the Board is as follows:

- to ensure that all relevant information on Brukunga is examined and analysed;
- to make appropriate recommendations to the Minister for Primary Industries and Resources SA on both short and long term strategies to deal with the pollution emanating from the Brukunga mine site and rehabilitate the mine site; and
- to facilitate consultation and the dissemination of information to stakeholders.

The Board will ensure that all appropriate records are kept as to expenditure and its purpose. The Board shall put broadly costed proposals for both short and long term remedial measures for Brukunga mine and any pollution arising from the former operation of the mine and its resultant consequences, to the Minister for his consideration. The Board will ensure that the Ministerial approved recommendations are acted upon. The Chairman undertakes to keep the Minister fully briefed prior to agreement on the Plan.

Getting solutions to pollution of Dawesley Creek
Draft environment improvement program (EIP)

- Required under licence with SA Environment Protection Agency.
- Short and medium term strategies to improve pollution of Dawesley Creek.
- Draft EIP submitted to EPA.

Dawesley Creek isolation study

- Proposal to pipe water in Dawesley Creek past the current mine site area.
- Aim of creek diversion is to retain pollution in the mine site portion of the creek to prevent further downstream pollution.
- Pump water from creek in mine site area and clean out creek bottom to remove heavy metals.
- Tenders for project for feasibility study being developed.

Assistance with downstream water supply issues

- Landowners have been consulted on their water requirements.
- PPK engaged to seek common solution to providing water to those landowners where required.
- Review will be completed in December.

CSIRO desktop review: Brukunga cattle infertility

- Review of data almost complete.
- Study aims to derive causes of infertility.
- Project completion 13 December 1999.

Study of extent of pollution along creek

- Project being undertaken by Primary Industries and Resources SA geologists and geochemists in consultation with CSIRO.
- Sampling program and analysis complete.
- Report due November.
- Additional study of heavy metal levels in pasture on the river flat to commence immediately.

Gap analysis

- Review of studies and reports on Brukunga and Dawesley Creek was undertaken.
- A review of the gap analysis report aims to determine priorities for further studies that might need to be undertaken to aid in developing strategies to overcome pollution.
- This study aims to determine whether we need to augment actions currently being undertaken and strategies for long term solutions.
- Draft report complete.

Use of biosolids on tailings dam

- To conduct a study into the use of neutralisation plant precipitate, dried biosolids and septic tank sludge, in rehabilitation work at the Brukunga mine site.
- The consultant is required to conduct a study of the physical properties and chemical composition of septic tank sludge, dried 'biosolids' and neutralisation plant precipitate and to appraise their application in mine site rehabilitation work at the defunct Brukunga mine site.

- Concern has been expressed that the level of cadmium and other heavy metals contained in the biosolids and neutralisation precipitate may be remobilised by erosion and leaching of the acid environment and, hence, make a significant contribution to the pollution level of site effluent.
- The study should consider the beneficial uses of the materials and determine their relative potential to erode, leach and release heavy metals to effluent.
- The study should provide a perspective on the relative significance of any heavy metals contribution to the total metal levels in effluent from the defunct mine site.
- Completion date March 2000.

Immediate improvements to acid drainage from mine site

- Primary Industries and Resources SA has studied water and drainage for the mine site and tailings dam area.
- Information of value in designing improvements to acid seepage interception.
- One water capture sump in mine site will be relocated and current wells deepened which will incur additional plant and pump operating costs, but will reduce acid seepage from the mine.

Rehabilitation of tailings dam area

- Covering of tailings dam area with soil and trees continues.
- Attempts have been made to intercept the artesian spring located under the tailings dam.
- This will reduce seepage through the tailings dam.

Ivan Dainis – Department of Premier and Cabinet SA

A conceptual model of the chemical loads in Dawesley Creek arising from acid sulfates in the Brukunga quarry

Abstract

A conceptual model is described which explains the seasonal behaviour of the load of acid sulfates and associated heavy metal salts that impact on Dawesley Creek at the Brukunga pyrite quarry. This quarry model is based on flow proportional monitoring in 1998 and also earlier historical data. It explains the variable iron and aluminium values observed and also provides tools for the identification and explanation of anomalous monitoring results. This new understanding of elemental variations and inter-relationships should simplify monitoring and assist the inter-correlation of biological testing results with those from chemical monitoring.

Introduction

Given the present speed of change and the need for flexibility, it is sometimes better to have the best definition of a problem rather than the best collection of answers. The present paper is about better definition of the situation at Brukunga. It has been developed independently of the Brukunga Remediation Board and also Primary Industries and Resources SA (PIRSA). Critical comment on the conceptual model presented is therefore most welcome, and particularly from the scientific community.

In specific regard to the core problem, the scientific and mining literature provides a plethora of general and also specific solutions to the problem of acid mine drainage (AMD), that is, to the environmental problems caused by acid sulfates that come from the oxidation of exposed mineral sulfides. These solutions include blocking oxygen access, reducing water access, inhibition of sulfur converting bacteria, chemical fixing or capture of iron sulfate product, in situ and post facto neutralisation, and also acceleration of oxidation. In the main these methods have involved physical containment of the problem as well as one or more specific techniques. So, in many reported examples it is difficult to quantitatively assess the claimed benefits of a particular method. This is particularly true where initial work has been reported but no long term follow up and evaluation has occurred to compare actual results against initial project claims.

In fact, the containment of sulfide oxidation in abandoned and sulfide-rich mines has some similarity to the containment of radioactive waste. Both involve continued monitoring, the long term alienation of affected land, and ever-present environmental and community concern. Also, there is often unavoidable government commitment and expenditure over what may be several hundred years with little or no economic return.

Accordingly, the definition of the core problem, and also community 'ownership' of it, appear central to any long term strategy that deals with the above problems. Today, however, long range planning appears unwelcome and unpopular. It is often difficult to forecast performance milestones, and particularly ones that satisfy expectations of speedy action and of zero risk. Moreover, such planning may be incorrectly perceived as a delaying tactic and an admission of indecision about a solution or solutions. However, it is salutary to consider that the costly and past solutions for Rum Jungle and Captain's Flat have not fully contained the respective problems. It therefore appears risky to assume that any solution is final until all oxidation and /or transport of acid sulfates has ceased or has been totally contained.

The modern approach to the above dilemma (and as embodied in the EPA licensing of sites) is that of a strategy of continuous improvement involving regular and open discussion of the costs and benefits of action(s). For Brukunga, such discussion, and also the further definition of the core problem[1], are continuing. Research into scientific and technical questions is also continuing. Much of this has been funded and greatly facilitated by the Managing Mine Wastes Project of the Australian Nuclear Science and Technology Organisation (ANSTO). Research results have been presented and discussed at two annual seminars.

Background

The first environmental study was conducted from 1973 to 1975 by the Australian Mineral Development Laboratories (AMDEL)[2]. It investigated the size of the problem, identified the major sources of sulfates, estimated chemical loads and oxidation lifetimes, and proposed further studies and possible solutions (including the present water treatment plant).

The second study by Environmental Geochemistry International Pty Ltd (EGI)[3] (and involving sub-contractors such as ANSTO) spanned 1992–1995. The study calculated a water balance and estimated sulfate production sources and annual loads within the mine site. It also reviewed the operation of the neutralisation plant, and suggested a range of remediation options with broad costings. In particular, the latter study showed that less than half the annual load of acid sulfates was being captured and neutralised at the water treatment plant. It also confirmed the earlier AMDEL finding that most of that load arises from the quarry's waste rock heaps. This corrected the previous conclusions by Dainis[4] and also Read[5]. Both these

[1] For PIRSA (SA) the author has identified several information gaps existing as at January 1999.

[2] Various and early AMDEL Reports are accessible via the SA Mines and Energy Department's Open File Envelopes Nos. 2463, 2582, and 2277. The latter two envelopes also contain three early 1973-75 Progress Reports whose data forms much of the basis for later AMDEL summary reports nos. 1015, 1065 and 1096. AMDEL Progress Reports from 1980 and up to 1991 deal with routine monitoring of creek water acidity.

[3] Rehabilitation of the Brukunga Mine Site and Tailings Dam. Final Report Volumes 1&2, March 1995. (Previous AMDEL and Engineering and Water Supply Department (E&WS) water chemistry data has been collated and graphed in Volume 2 of EGI's October 1993 Interim Report).

[4] Dainis I, Environmental Chemistry at Brukunga. Report to the Director-General, SA Department of Mines and Energy. December 1992.

[5] Read R E, SA Department of Mines and Energy Report Book 86/83.

workers independently and erroneously concluded that the main load derives from the quarry benches and cuts.

Before the AMDEL study, major environmental complaints had been made by land owners on Dawesley Creek about high summer loads affecting their stock. Accordingly, subsequent drainage, collection and pumping works from 1974 to 1978 centred on reducing mine site releases of acid water during summer months. A considerable reduction was achieved, but the resulting storage/evaporation pondage on the tailings dam proved unstable and inadequate.

A water treatment plant to neutralise collected seepages was therefore commissioned by the South Australian government in 1980. At that time the plant was planned to be low-cost, and automated, and to only require one operator on a part time basis. It was also projected to be redundant by the year 2000 because it was hoped that remediation solutions would by then have contained or greatly diminished the annual load into Dawesley Creek. It was however, made subsequently clear[6] by government that while the water treatment plant coped with general seepage flows it could not be expected to perform adequately during exceptional flow and/or flood episodes. This point was reiterated in 1995 by Miller in the EGI Report[7].

Dawesley Creek has only a small catchment on the eastern scarp of the Mount Lofty Range, and its annual flow is considerably augmented by treated water from the Woodside sewerage treatment works upstream. Rainfall is on average about 600 mm per year and it can be erratic, varying by as much as 50% about that mean. Figure 1 shows the recorded rainfall at Brukunga for the period 1974 to 1998 inclusive (all figures are located at the end of text). Net evaporation over rainfall is high and can exceed one metre a year. The result of early AMDEL studies is shown in Figure 2[8]. Evaporation is only exceeded by rainfall during the winter months.

It is perhaps no coincidence that major complaints about creek pollution have occurred either at times of low creek flow or of high rainfall and/or flooding. The AMDEL study was created because of landowner concerns during 1972 when rainfall was unusually high. Similarly, there were complaints during the record 1992 floods. That year also saw a complaint about the rise of sulfates in soils under and around houses at Brukunga, and which was the subject of a very extensive investigation by the South Australian Housing Trust. That such flooding may re-occur is exemplified by the 114 year-old rainfall record for the nearby town of Nairne as shown in Figure 3.

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[6] In his letter of 1st October 1981 to the South Australian Water Resources Council the Director-General, Department of Mines and Energy states that '...whilst the (neutralisation) plant will alleviate the problem significantly, it will in no way completely remove it.'

[7] At page xii in Vol 1 Miller states: "There will be flow events where high contaminant loads are leached from exposed waste rock areas which would by-pass the collection system resulting in short term non-compliance with water quality targets in Dawesley Creek.'

[8] Data from AMDEL Report 1015. November 1974.

Figure 4 shows the close similarity of Brukunga's rainfall with the *ca.* 20 mm higher Nairne rainfall. Although there is a known regional trend to lower rainfall over the last thirty years, the regional record indicates that flooding can be expected at Brukunga at least once every ten years.

Chemical monitoring

An interim October 1993 report within the EGI study summarised all the analysis data to that date on creek and Minnesota water samples. This contains limited data on metals. Most reported data is periodic acidity measurements, namely pH and acidity as calcium carbonate equivalents. The latter values were converted to sulfate values by Stuart Miller of EGI for his study on annual sulfate production. It is fair to say that up to 1996 monitoring was sporadic[9], and, except for the above major investigations, did not provide information that could be used confidently to develop a model of the quarry.

Following the licensing of the site by the SA Environment Protection Agency (EPA), chemical and biological monitoring was stepped up by SA Water in 1996 and has been maintained by Primary Industries and Resources SA (PIRSA) since it took over management of the mine site in 1998. In particular, flow proportional sampling through automated stations just above and below the mine site commenced in February 1998. The data resulting from this work, as kindly supplied by Tim Thompson of AWQC and PIRSA officers, has made the current conceptual model possible.

The conceptual model

The model and annual cycle described below **does not include the tailings dam.** It is implicitly assumed that in 1998 all the tailings dam seepages were neutralised and only clean water was returned to the creek. It is also assumed that the rate and scale of sulfide oxidation in the quarry is invariant. Also, since the volume of groundwater (i.e. artesian water) entering the quarry from outside the mine site is relatively small, and its chemical load negligible, it has been ignored in the model.

It is also emphasised that the present model is based largely on 1998 chemical results on creek water samples taken **immediately** below the mine. Sulfate and particularly metal concentrations decrease with increasing distance from the mine site, and are affected by incoming stream chemistries and also existing sediments. The downstream effects such as dilution and neutralisation and flocculation-sedimentation change the concentration profile of the various elements, and this is discussed later.

[9] This does not include the important E&WS surveys of the Bremer River catchment.

The core of the model (and annual cycle) as it applies to the Brukunga **quarry** involves the following:

1. At the quarry there is constant generation of acid sulfates whose transport into Dawesley Creek (and therefore impact) is critically determined by the rainfall and drainage patterns at the quarry;
2. Although there are constant seepages into the creek, with the highest concentrations of sulfates occurring during low-flow (i.e. summer months), the creek receives its greatest sulfate (and metals) load in winter when dilution from creek flow is at its highest;
3. The chemical makeup of the sulfate load changes seasonally in character from summer to autumn and from winter to spring because of an interrelated mix of several factors, including:
   - the major attack by ferric sulfate on sodium feldspar (albite), resulting in the formation of aluminium sulfate with a corresponding decrease in the observed iron load (and also in the utility of iron as a direct and quantitative indicator of AMD);
   - the accumulation, through 'wicking' and evaporation, of aluminium and iron sulfates at the quarry (with the exclusion of other metal sulfates, such as manganese and zinc sulfates, through natural crystallisation effects);
   - the formation and residence times of ferrous and ferric sulfates within the quarry's various zones;
   - the differing water penetration and leach times at the various sulfide oxidation zones, which influence the differing sulfate transport mechanisms and rates within the quarry; and
   - the formation of sparingly soluble basic sulfates, mixed salts, and hydrated oxides as secondary mineral deposits.
4. Generally, the higher the concentration and load of total sulfate the higher the concentration of iron, aluminium, cadmium, manganese, zinc, and nickel;
5. The concentration ratios between manganese, zinc and nickel are relatively constant (and therefore very useful for downstream studies and for internal checks on monitoring results);
6. Because iron, and to a lesser extent aluminium, 'drop out' with increasing distance from the quarry, the concentration ratios between these elements and the other major elements are more divergent (and consequently less predictable);
7. The 1998 cadmium concentrations are anomalous (in fact, a threefold increase in surficial cadmium is indicated from past cadmium to zinc ratios, and also from observed 1998 transport characteristics–see later); and
8. There is a close relationship between total sulfate content and 'total iron' content (the latter being the calculated sum of the iron and half the aluminium contents).

Figures 5–29 illustrate these particular characteristics. They can also be used to examine and predict future behaviours.

In the examination of figures 5–29 it needs to be borne in mind that the primary data comes from the analysis of unfiltered samples of creek water and contains a number of outliers (i.e. analysis values that appear much too large or too small relative to the observed norm). The very fine particles within these samples can contain some of the elements of concern[10]. Small changes in sampling methodology and sample storage can significantly affect the aggregation and settling of such particles.

Because 1998 was the first year of automated flow proportional sampling the analytical data shows considerable 'noise' and also some large variations in results. Nevertheless, some variation can be expected in environmental water samples, particularly at low levels of measurement, and also in samples having high dissolved solids content and, therefore, greater density and capacity for suspension of fine sediment. There is also some 'noise' in the volumetric measurements of flow in Dawesley Creek. At the time of reporting the analytical data the gauging equipment had not been calibrated. Now that this has been done it is known that the 1998 flow results are about 10% too high. However, the results shown in figures 5–29 have not been corrected because such adjustment does not affect the present model and related findings.

The following data treatment has used pattern analysis, mainly through the plotting of cumulated loads i.e. summed time series, and also the graphing of analysis values. Here, the main focus has been the visualisation of seasonal changes over time, rather than maximum values and their environmental significance in terms of the Australian water quality guidelines. In the summed load figures, the primary load data of Thompson was used, as supplemented by calculations on data supplied in early 1999 by Water Data Services Ltd to PIRSA. The concentration figures also used the latter data source.

Transport of sulfates and metals from the quarry

Figure 5 shows the cumulative 1998 rainfall profile at Brukunga compared with the 1998 cumulative flow profile for creek water leaving the quarry. Figure 6 shows the comparison between leaving and entering profiles. The difference between these profiles is largely due to the rainfall over the mine site catchment. In 1998 the additional flow only became pronounced after day 200 (i.e. in July). Because of Brukunga's erratic rainfall, the onset of this flow difference (as due to rain at the mine site) will vary from year to year. In general, and from an examination of past rainfall profiles (see later) the onset can generally be expected in the period April –July[11].

[10] During 1998 one water sample drawn on 22/4/98 was mistakenly filtered. This filtration reduced the iron concentration one hundred fold, and also halved the sulfate concentration. Other element values including aluminium were not significantly affected. This suggests that the main suspended sediment may be a basic ferric sulfate such as natrojarosite, or a basic sulfate of iron and magnesium as noted by AMDEL previously.

[11] This assumes that the flow contribution (of about 50ML 'return' water) per year from the water treatment plant is relatively uniform during the annual cycle.

Figure 7 shows the profile of cumulative flow compared with the scaled cumulative sulfate load profile. Both profiles are very similar after winter but what is especially noteworthy is the early separation of profiles which coincides with the first significant rain of the year **before** major flows occur in the creek. An examination of the early AMDEL data as reported by EGI Ltd[12] shows that an early rise, a dip, and then a larger rise in sulfate concentration is a general Brukunga characteristic for the first half of the year. This rise can now be attributed to the autumn 'flush' of the 'surface' sulfates that have accumulated during summer. This 'flush' is rich in aluminium sulfate: a result of several factors, including the increased contact-reaction time of ferric sulfate with albite in the summer months, the greater tendency of iron (relative to aluminium) to form basic sulfates and hydroxides, and the filtering effect of decomposed rock. As might be expected, the evaporative capture of sulfates is more evident on the iron-rich quarry benches and cuts than on and in the waste rock heaps. Nevertheless, it is also during summer that the waste heap seepages have their greatest concentration of sulfates.

Figure 8 compares the cumulative load profiles of sulfate and aluminium. The aluminium load profile exhibits the initial 'flush' at the first rains, another 'flush' as the first rains take time to affect the seepages in the waste heaps, and then a tailing of this into the main load as a consequence of consistent winter rain. This profile also shows the plateauing of the aluminium load in late winter. This is followed by a relative increase in aluminium as seepage volumes shrink and surface evaporation increases.

A scaled comparison of sulfate and manganese load profiles is shown in figure 9. In fact the cumulative load profile for manganese is very similar to that of the cumulative rainfall profile shown in figure 5 above. The explanation for this is that this element's release and transport is least affected by pH changes and secondary product formation.

Figure 10 shows the scaled cumulative load profiles for sulfate and also various metals and, therefore, the varying 'transport curves' for 1998. The profiles for both zinc and nickel are closer to that of manganese than they are to both iron and aluminium. In particular, the nickel and zinc profiles are intermediate between the rainfall and creek flow profiles. In fact, a plot of nickel values against rainfall indicates an approximate relationship between rainfall and nickel values. This is what can be expected from nickel's chemical behaviour and particularly its avoidance of entrapment in crystalline sulfate deposits and evaporites. Aluminium's load profile is similar to sulfate's. This is not surprising since aluminium sulfate is the main sulfate transported out of the quarry into the creek. Iron's profile differs from that of aluminium and also the profiles for manganese, zinc, and nickel. This important aspect is discussed more fully below.

Figure 11 specifically contrasts the **scaled** cadmium profile with that of zinc. Cadmium exhibits anomalous behaviour here since it is expected that cadmium

[12] See footnote 3.

should parallel zinc. Cadmium sulfide occurs as a minor impurity in zinc sulfide and is released upon the latter's oxidation by ferric sulfate[13]. Early AMDEL data showed the cadmium to zinc ratio to be relatively small and constant. However, the 1998 results show a considerable increase in this ratio.

Historical comparison of chemical loads

Recently Thompson has compared the monitoring data obtained from 1996 to 1998 with past E&WS data from Bremer River surveys[14]. His plots of historical median, minimum and maxima data as weighted for volumetric flows have shown that recent pH values and concentrations of heavy metals are similar to those seen in the 1970's and 80's. Table 1 shows the comparison between the early creek pollution load calculated by AMDEL with that calculated in 1998 by Thompson and as used in the present figures. In examining this comparison, it needs to be borne in mind that the neutralisation plant was installed in 1980 and had got rid of the tailings dam 'backlog' by 1987. Also, a collection system for mine site seepages, which was termed 'partial diversion' and which claimed to reduce acid levels in the creek by 33–50%, was implemented from 1976 onward, and before water treatment in 1980.

Accordingly, the observed 1998 load is that **solely** due to uncaught seepages. This is in contrast to the larger 1973–1974 load which comprised a complex mixture of tailings dam seepages with 'aged' and concentrated contents from the holding pond plus the uncaught seepages as at present.

Rainfall in 1998 was below average whereas the period 1973–74 saw unusually high rainfall and also a pollution plume extending down to Lake Alexandrina. The size of that plume was apparently due to the unauthorised release of acid from the holding pond at the mine site. Both events suggest that the 1973–1974 AMDEL gaugings were above what may be considered as average.

A comparison of the iron and aluminium loads in Table 1 suggests that the total sulfate load in 1973–1974 was double the load observed in 1998. That high sulfate level accounts for the higher load figures for manganese, zinc, nickel and cadmium. Nevertheless, and for the reasons given above, Table 1 does not negate Thompson's conclusion, nor an earlier 1992 postulate[15] that the amount of 'uncaught' pollution in the creek has not decreased. Further evidence of a cadmium anomaly is presented by the differences in two pairs of figures. Figures 12 and 13 show the observed relationships between zinc and creek flow, and also between zinc and sulfate concentration. Figures 14 and 15 show the corresponding data for cadmium. Whereas zinc displays the expected fall-off in concentration with increased flow, cadmium does not. Also, while zinc shows a regular increase with increase in sulfate, cadmium displays a more random 'shotgun' pattern.

Table 1. Comparison of metal loads in Dawesley Creek between 1973-74 and 1998

[13] Fowler T A and Crundwell F K, Applied and Environmental Microbiology, 1998, **64**, 3570–75.

[14] Thompson T , Graphs as presented at the November 1998 Brukunga Workshop.

[15] Dainis I, A Proposal for Measuring the Present Impact of Brukunga Acid Quarry Effluent on the Bremer River System. An April 1993 proposal to the Brukunga Steering Committee.

Metal	1973-74 Load (tonnes In 12 Months)[16]	1998 Load (tonnes In 10 months)	Comment
Iron	200	59	A very significant decrease, which reflects the success of the water treatment plant in dealing with the iron-rich seepages from within the tailings dam[17].
Aluminium	150	138	A similar amount, and one that reflects the major contribution made by the waste rock heaps (as noted by AMDEL and EGI these heaps in comparison with the tailings dam and quarry benches 'bleed' much more aluminium sulfate than iron sulfates).
Zinc	20	4.1	A significant reduction, and one that probably reflects the relatively low concentration of zinc sulfide in waste rock when compared to the orebody and tailings.
Manganese	15	5.3	Ditto for the manganese dioxide in waste rock
Nickel	1	0.26	Ditto for the nickel sulfide in waste rock
Cobalt	1	-	No comment
Copper	0.5	0.17	Parallels reduced manganese load
Cadmium	0.1	0.06	Reduced load relative to 1975 estimate. However, the ratio of cadmium to zinc has trebled.
Chromium	-	0.04	No comment

The nature of this divergence between cadmium and zinc suggests that the 'extra' cadmium comes from a surficial source or sources. The most possible source is the large tonnage of sewage sludge and septage which has been used for 'greening' the mine site over recent years. Here it needs to be noted that surface water from the capped tailings dam is channelled to the creek without treatment. Most of the sewage sludge used at Brukunga has been used to vegetate that surface.

[16] The AMDEL estimates are based on an annual creek flow of 800 ML, whereas the 1998 flow, as measured for 10 months, was 781 ML.
[17] The tailings dam seepage is essentially a solution of ferrous sulfate saturated with gypsum.

Another more recent source of cadmium could possibly be dumped gypsum sludge from the water treatment plant. This contains metal hydroxides that could be mobilised, especially as emplaced within the quarry's very acid southern extension and above the original creek bed. However, this source can be discounted because acid water would also be expected to mobilise zinc, thus leading to little overall change in the cadmium to zinc ratio.

The iron to aluminium relationship

A notable and puzzling feature of data from Dawesley Creek has been the seemingly erratic behaviour of iron. This can now be explained by two main factors: the very acid[18] and reactive nature of ferric sulfate in an aqueous environment and the penetrative and major transport effects of winter rain. AMDEL observations twenty five years ago (and as embodied in the present model) were that:

- the chemical composition of seepages and flows from the various parts of the quarry are remarkably constant irrespective of rainfall, and in general the seepages from quarry benches and cuts are iron-rich;
- there are delays of up to six weeks between high rainfall episodes and increased seepage flows from the waste rock heaps[19];
- the main constituent of the latter flows is aluminium sulfate; and
- most of the annual pollution in the creek arises from waste rock seepages.

Figure 16 from 1998 data shows the seasonal change in the ratio of aluminium to iron in the creek below the quarry. There is an appreciable difference between summer and winter as shown by the polynomial trend line. This effect is discernible in all historical data examined and is explained by the increasingly penetrative effect of winter rainfall on active oxidation zones. This especially facilitates transport of iron sulfates from quarry faces and benches into the creek. This pronounced rise in iron content occurs while there are increased (and aluminium-rich) seepages from the waste rock heaps. In the smoothed 1998 data there is a definite two-fold variation in iron to aluminium between winter and summer flows. In regard to individual flows there is as much as a five-fold variation. Such a large variation may owe much to two factors: less iron precipitation in winter and greater elution of sulfates from iron-rich, and active, oxidation zones. This seasonal variation in iron and aluminium concentrations is very significant in terms of visual and also biological impacts downstream from the quarry. Mr Peter Schultz of AWQC deals with the latter impact in his paper on biological monitoring of Dawesley Creek.

The aluminium sulfate at Brukunga comes from the decomposition of the mineral albite by ferric sulfate. An examination of the 1998 data suggests that there is an overall loss of one mass of iron for the gain of two masses of aluminium. To test that hypothesis the observed 1998 aluminium values were halved to give 'iron equivalents' and this result was added to the 1998 iron values to give a measure of

[18] In water ferric sulfate is a loose combination of polymeric ferric hydroxide and sulfuric acid.

[19] Rain in late summer/early autumn may also be delayed in its effects because of stratification. Clear fresh water lying above red sulfate solutions can often be seen in the quarry, and particularly after rain in late summer.

'total iron', that is, the iron values that would theoretically have been observed in Dawesley Creek as if there had been no attack on albite.

Figure 17 shows that the 1998 cumulative load profile of 'total iron' can be superimposed on the scaled cumulative load profile of total sulfate. These and previous figures confirm that the Brukunga quarry is a heap leach system basically driven by iron sulfate generation, with subsequent sulfate modification before the expression of these sulfates through seasonal rainfall and water transport mechanisms.

Relationships between other elements[20]

In general, high values of sulfate (and aluminium) are accompanied by high values for other elements. Figures 18-21 show 1998 plots of various elements against sulfate. Cadmium (figure 15) is anomalous. When obvious outlier values are ignored it is possible to obtain the following relative abundances (sulfate at 1000).

Aluminium	65	i.e.	Al:sulfate = 65:1000
'Total Iron'	110	i.e.	'total Fe':sulfate = 110:1000
Manganese	7.5	i.e.	Mn:sulfate = 7.5:1000
Zinc	4	i.e.	Zn:sulfate = 4:1000
Nickel	0.3	i.e.	Ni:sulfate = 0.3:1000

AMDEL workers had previously pointed out that manganese is a good reference element, and this is borne out in figures 22 and 23 which show that a direct relationship exists between manganese values and those of zinc and nickel. The relative abundance of the latter two elements relative to manganese (as 10) is as follows:

Zinc	6	i.e.	Zn:Mn = 6:10
Nickel	0.4	i.e.	Ni:Mn = 0.4:10

Figure 24 shows a plot of nickel against zinc, the general ratio being 1:15. The above empirical relationships are invaluable for verifying monitoring data and for the detection of sample errors and outlier values. In particular, the relative ratios of the three elements, manganese, zinc, and nickel, are useful for the study of downstream dilution effects.

A 'typical' year at Brukunga

The annual calendar cycle at the quarry generally begins in January and February with an intensification of summer evaporation. Waste heap seepages may diminish depending on what rain has fallen in November and December. Sulfate solutions are concentrated and are 'wicked' to rock and soil surfaces where they form

[20] It needs to be noted that the rock in waste heaps and beds has an overall lower content of elements such as manganese, zinc, cadmium, and nickel when compared to rock in the orebody below the quarry benches and on exposed quarry walls. Accordingly, the proportions of these elements in the creek can be expected to vary, depending on the relative contributions via seepages from each type of rock. Any derived and empirical ratio between elements therefore rests on the assumption that these contributions are relatively constant from year to year.

efflorescences and crusts of mixed aluminium, iron and magnesium[21] sulfates. The ferric sulfate from continuous sulfide oxidation actively reacts with albite to give aluminium sulfate product. It also leaches manganese, zinc, cadmium, nickel and other minor metals from their constituent minerals. The crystallisation of the mixed major sulfates results in a winnowing out of lesser metal salts, and these remain in solution and are expressed in the concentrated seepages.

Surface quarry flows are virtually non-existent in summer and the creek water contains very little iron and much aluminium sulfate. Because of the low flow (as dependent on upstream releases) the sulfate concentration may be very high. However, the actual chemical load, and particularly the sediment load, is very low relative to the coming winter months.

When the first serious rains fall in March and April (or as late as May) there may be a temporary dilution of the high sulfate concentrations in the creek, but usually there is an upsurge in sulfate concentration resulting from a flushing of quarry surfaces and efflorescences. After this initial surge there is a relative decrease while creek flow increases and rain extensively percolates into the waste dumps. From June to August, heavy and continuing rain also percolates into the quarry benches and reaches active oxidation zones to leach out increasing amounts of iron sulfates. As winter rain continues, the seepages from the waste heaps also become more iron-rich. At this stage the sediment and chemical load in the creek is at its highest, particularly if heavy falls result in major sheet flows across quarry benches and cuts. As the rainfall diminishes after August-September (or as late as November) the iron-rich seepages from the quarry also fall. The creek flow diminishes, the sulfate concentration rises and the aluminium-rich seepages from the waste heaps reassert themselves. It is early summer and the annual cycle starts again.

Accumulated sulfate

For the whole mine site the EGI report indicated an annual production of about 3,000 tonnes of sulfate per year, with the capture and neutralisation of about 1,200 tonnes per year. Thus, about 1,800 tonnes were reckoned as left uncaught. No estimate was made as to how much sulfate accumulated on the site, although it was stated that residual iron sulfates would providentially be converted to insoluble secondary products such as nitrojarosite. No comment was offered at that time on the likely fate of residual aluminium sulfate.

Aluminium sulfate is known to form sparingly soluble basic sulfates such as alunite; however, the low potassium levels and also very acid conditions at Brukunga would appear to inhibit aluminium removal through secondary product formation. In 1998 the chemical load of sulfates in Dawesley Creek below the mine site was about 800 tonnes, so at first glance it may appear that at least the same amount (i.e. 800 tonnes) accumulates each year in the form of soluble sulfate and also secondary

[21] Magnesium sulfate and calcium sulfate are the third and fourth commonest metal sulfates in the quarry respectively. However, little data exists on magnesium levels. Calcium sulfate is usually present at saturation levels, at which it creates problems through deposits and pipe blockages.

products (with the latter formation depending on the chemical nature of the metal). However, the observed variation in the aluminium to iron ratios, and also the shape of the cumulative load profile for aluminium, could be read to indicate that as much as 30% of the aluminium sulfate that entered the creek in 1998 was 'accumulated' sulfate[22]. That supposition emphasises the indicative and approximate nature of the sulfate estimates made in 1995. A better definition of the sulfate quantities involved at Brukunga should emerge from the continuation of present chemical monitoring when this is allied to annual water balances over the mine site.

Analysis of downstream elemental concentrations

The above conceptual model provides a picture of what leaves the mine site and how the initial chemical load is formed and transported, but it does not explain what happens to that load downstream. To do this fully, would need flow proportional sampling at more sites. Nevertheless, it is possible to gauge broad changes to that chemical load from the elemental and other analyses on creek water in 1998 from various downstream sampling points.

The concentrations found for each of the four elements manganese, zinc, nickel, and cadmium were plotted as a function of time at each of five locations. Figure 25 shows the scaled concentration of these elements immediately below the mine site. The trend for manganese is highlighted here, and also in figures 26 and 27 which show the elemental concentrations for locations 5 and 8 below the Nairne Creek and Mount Barker Creek junctions respectively. The trend lines for these downstream locations show that in 1998 the concentrations of the four elements **increased** in the middle of the year (i.e. in winter). This is in marked contrast with the **decreasing** trend in the concentration of these metals at locations 2 and 3 below the mine site.

Now, it is known that both Nairne and Mount Barker Creeks contribute very little of the above four elements. So, although there is a decline in overall metal concentrations due to dilution from these creeks when compared with just below the mine site, the observed increase in concentrations during winter at the lower locations indicates that fine transported sediment is a significant contributory factor in that increase.

Increased creek flow has previously been noted by AMDEL workers as responsible for the spread of metal-rich sediments from Brukunga to the lower Bremer River system. In fact, one long term consequence of the Brukunga mine site has been the inexorable accumulation of metal-rich sediments in the Bremer River system. From the 1998 data it is therefore possible to say that manganese, zinc, nickel and cadmium are at their highest observed concentrations below the Nairne Creek junction when least expected i.e. at the first 'flush' of the quarry and during heavy winter flows.

[22] Conversely, and relative to earlier years there may be greater conversion of ferric sulfate to aluminium sulfate in the quarry because of the accumulation of porous gossan and therefore greater retention of rain and sulfates.

Figure 28 shows the plot of cadmium concentrations at various locations during 1998. In particular, the data used for this graph indicates that in 1998 the Nairne Creek diluted the cadmium concentration threefold and Mount Barker Creek over 20-fold–an overall dilution rate of about 60. In effect therefore, and at worst, a cadmium concentration of 0.6 mg/L leaving the quarry in 1998 translated to an 0.01 mg/L concentration in the Bremer River below Mount Barker Creek.

Summary

The Brukunga quarry is a 'leaking' chemical reactor whose environmental impact varies seasonally according to evaporation, rainfall and Dawesley Creek flow. In particular, and contrary to what is generally the case in non-Mediterranean climates:

- The greatest load impact on the Bremer River system occurs at maximum flow i.e. during winter;
- Increased flow results in transport of sediment and therefore a winter rise in heavy metal concentrations downstream, despite the overall dilution in such concentrations; and
- There is extensive accumulation of aluminium and other metal sulfates during summer, resulting in a large 'flush' of these solutes when late summer/early autumn rain arrives.
- There appear to be regular and useful relationships as between total sulfate load and 'total iron' (i.e. the sum of iron and half the aluminium contents/values), between total sulfate and the elements manganese, zinc, and nickel, and also between the latter three elements themselves.

The major conclusion is that the variations in monitoring results are largely cyclical and can be linked to the annual rainfall profiles at the mine site. As shown in figure 29 there are significant differences in this profile from year to year. In fact, over the last 25 years no annual profile is the same. This means that a larger than average rainfall (such as occurred in 1992) will give proportionately larger chemical loads and associated downstream sediments in Dawesley Creek. Also, the longer the interval between the quarry 'flushes', the greater the likelihood that these events result in a high load impact.

As regular monitoring progresses it should be possible to improve the above model so as to predict the impacts for a particular rainfall, including the related impacts further downstream in the Bremer River. Also, it should be possible to achieve the under-riding objective of the above work–to determine whether observed changes in monitoring results are significant indicators of remediation performance, and whether the acid problem is declining, static, or increasing–and perhaps by how much.

Acknowledgment

This work would not have been possible without the welcome support of Messrs Ian Kowalick and Eugene Biganovsky, and also the assistance of Messrs Peter Grindley, Tim Thompson, Peter Schultz, and Ray Cox.

Figure 1
Brukunga Rainfall Annual Totals (1974-1998)

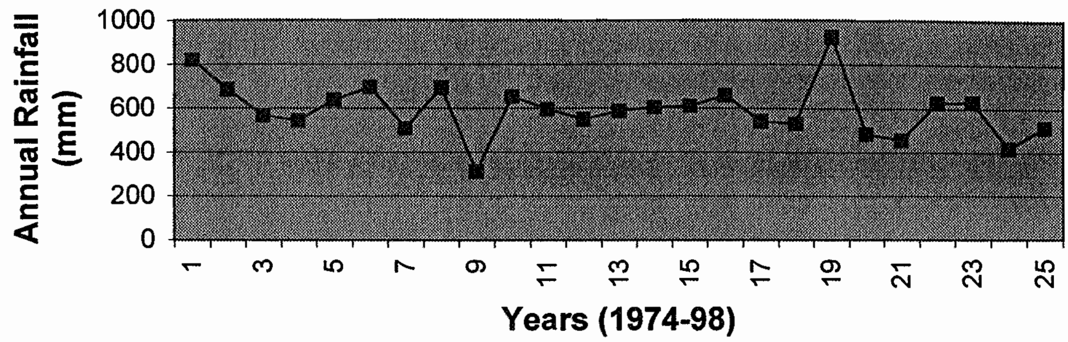


Figure 2
AMDEL Evaporation Study Results

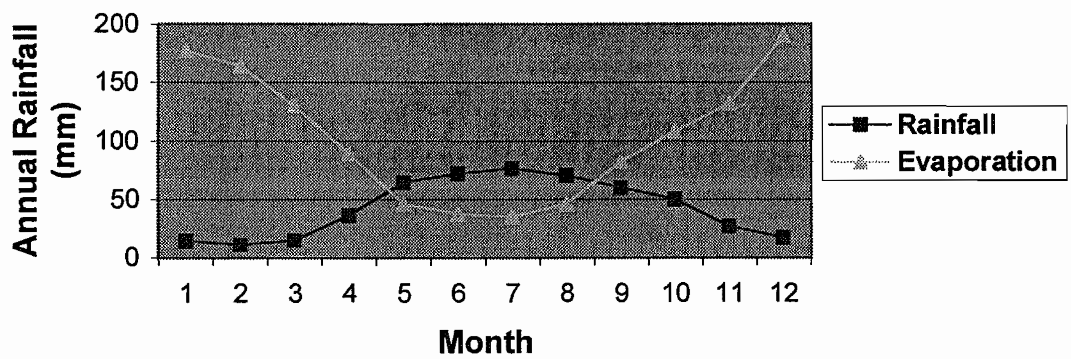


Figure 3
Nairne Rainfall Annual Totals (1884-1998)

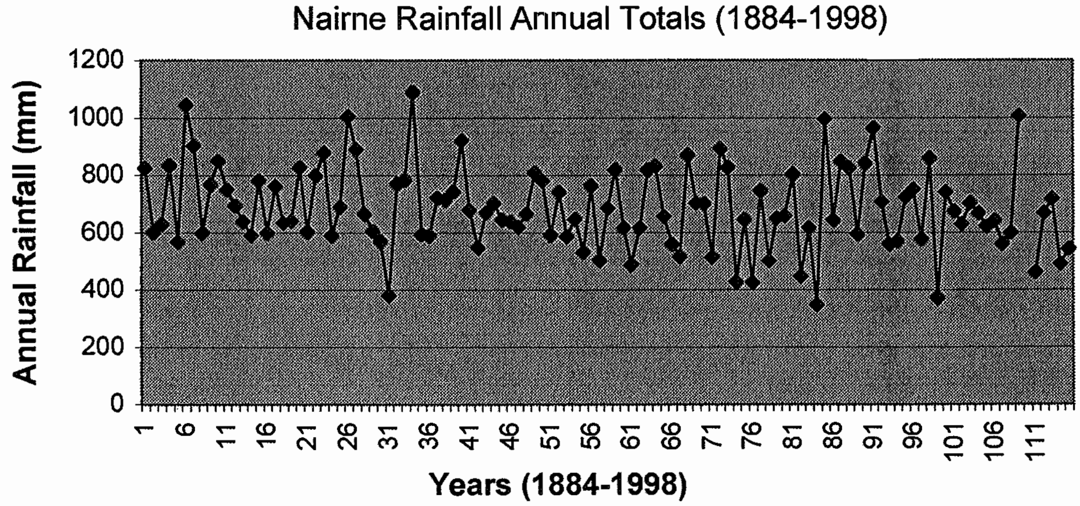


Figure 4
Nairne and Brukunga Annual Rainfall (1974-1998)

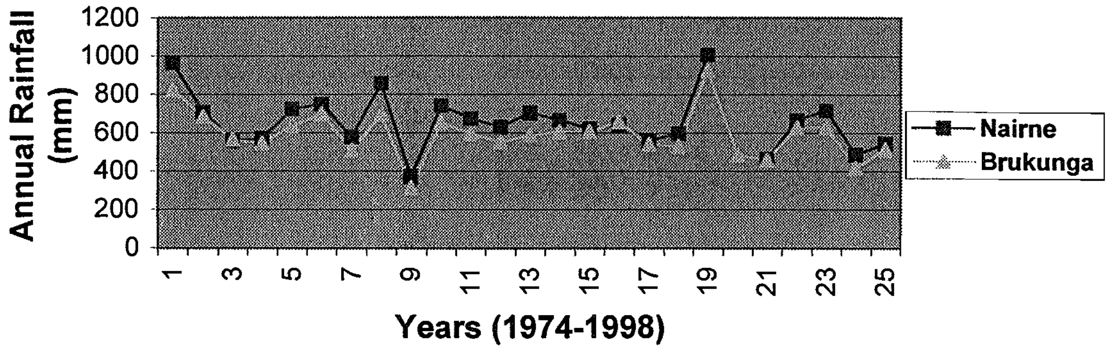


Figure 5
Cumulative Flow and Rainfall Profiles 1998

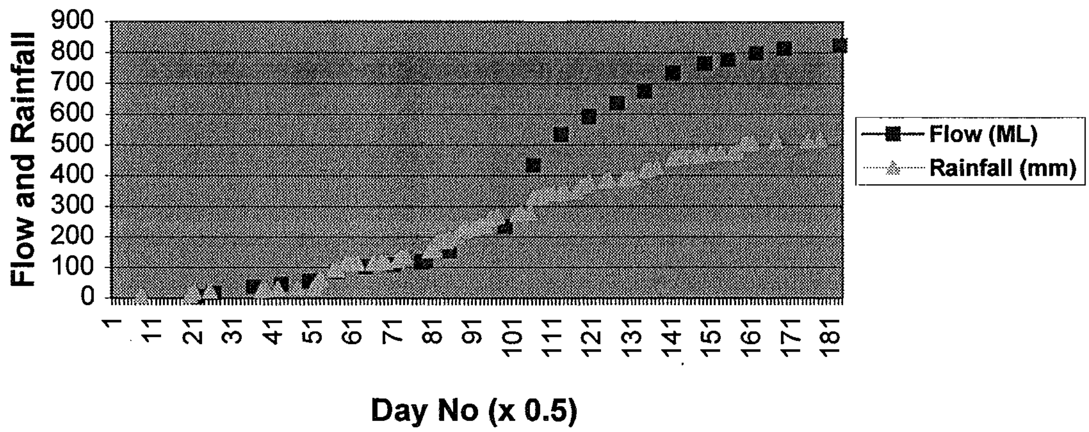


Figure 6
1998 Before and After Mine Site Cumulative Flows

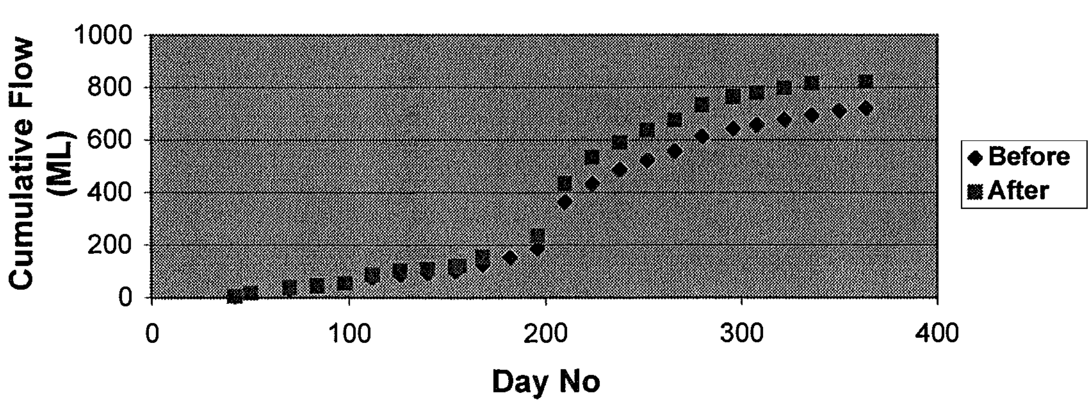


Figure 7
1998 Profiles of Cumulative Sufate Load (kg/100)
and Cumulative Creek Flow (ML)

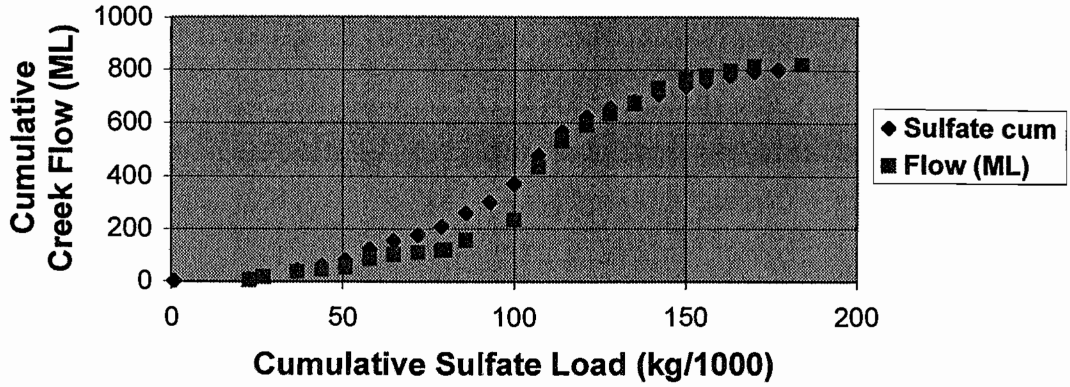


Figure 8
1998 Profiles of Cumulative Loads of Sulfate (kg)
and Aluminium (kg)

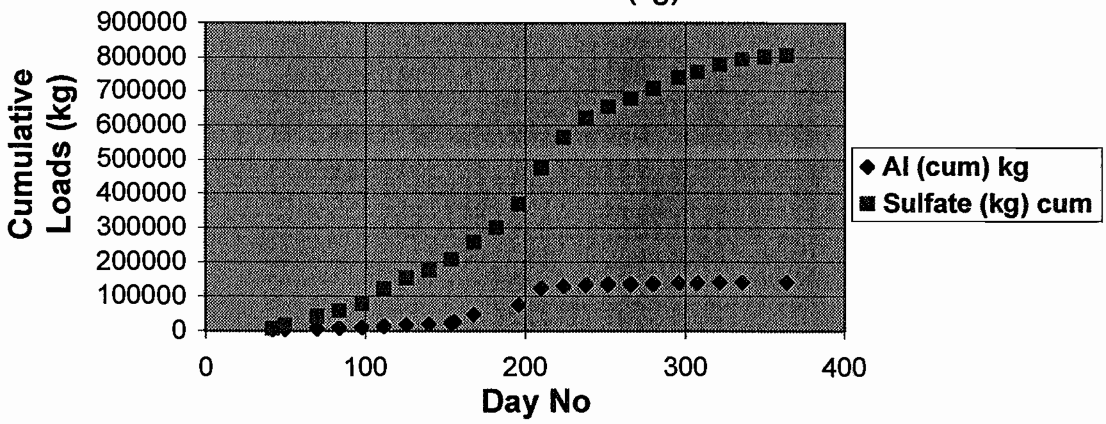


Figure 9
1998 Profiles of Manganese Cumulative Load (kg)
and Scaled Cumulative Sulfate Load (kg/100)

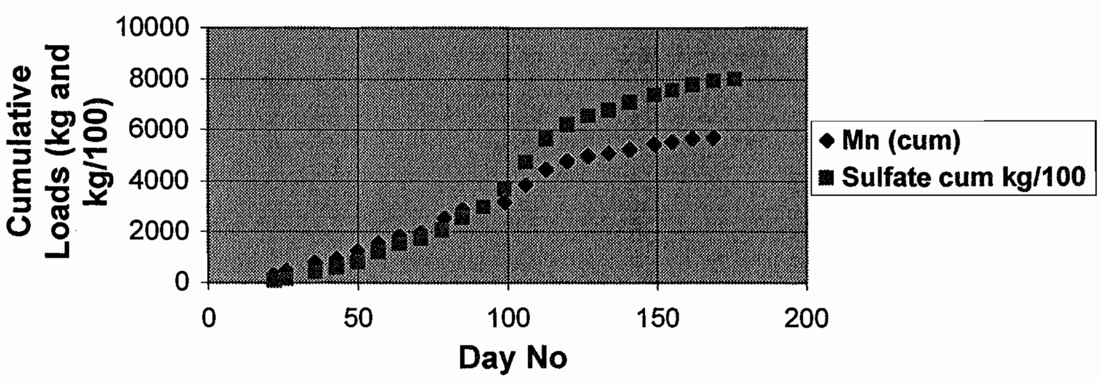


Figure 10
1998 Profiles of Scaled Cumulative Loads of Manganese, Zinc, Cadmium (x100), Nickel (x15), Iron (kg/10), and Sulfate (kg/100)

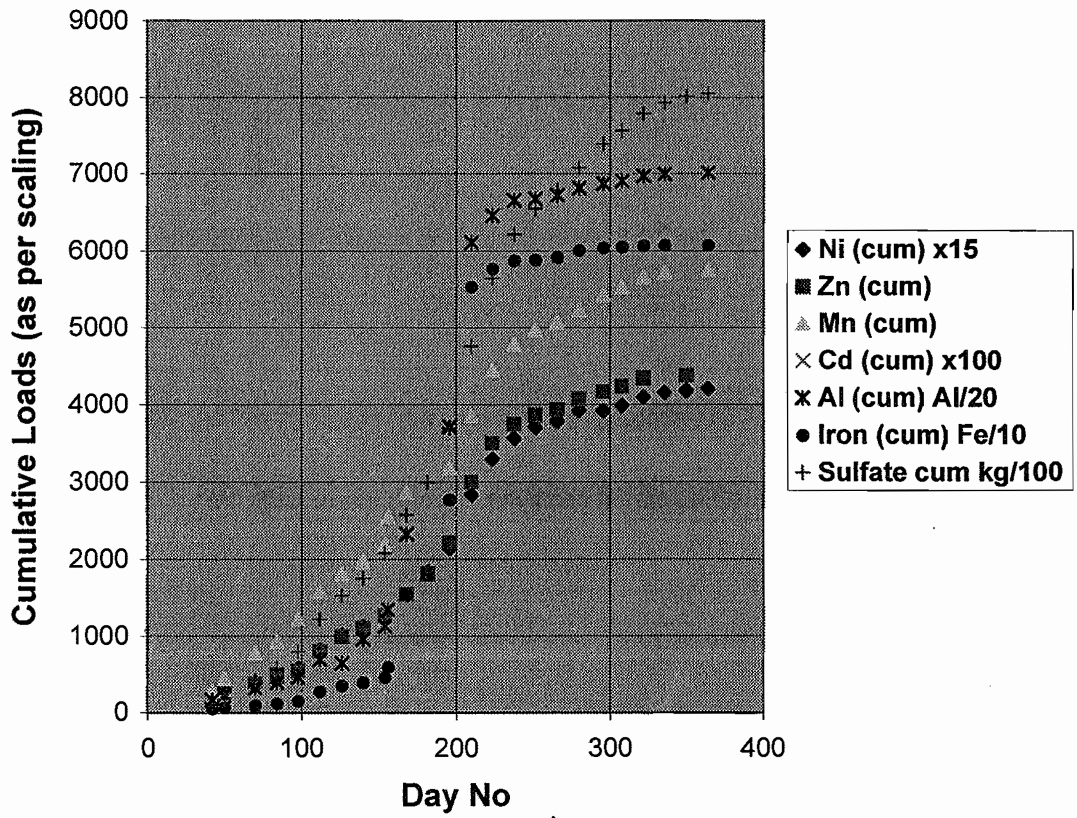


Figure 11
1998 Profiles of Cumulative Loads of Zinc (kg) and Cadmium (kgx100)

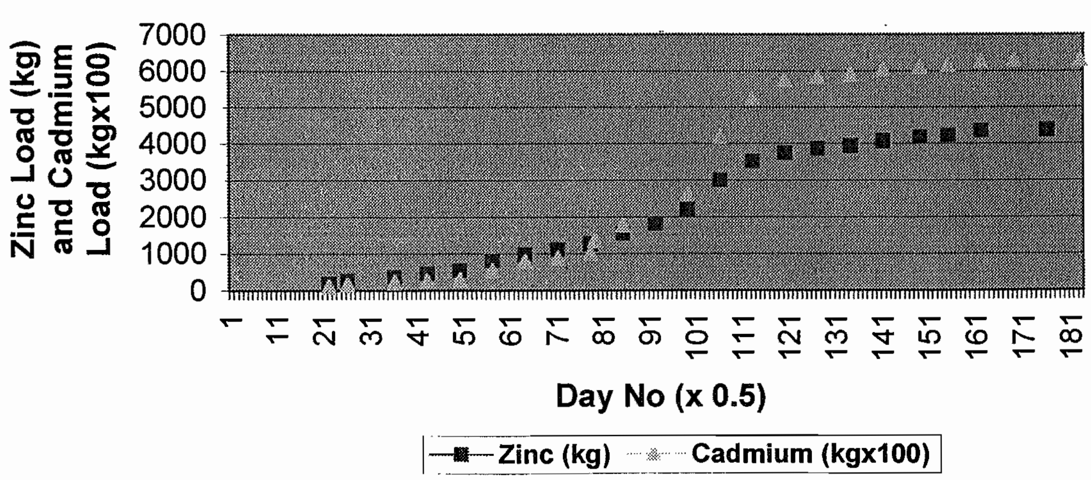


Figure 12
1998 Flow (ML) versus Zinc (mg/L)

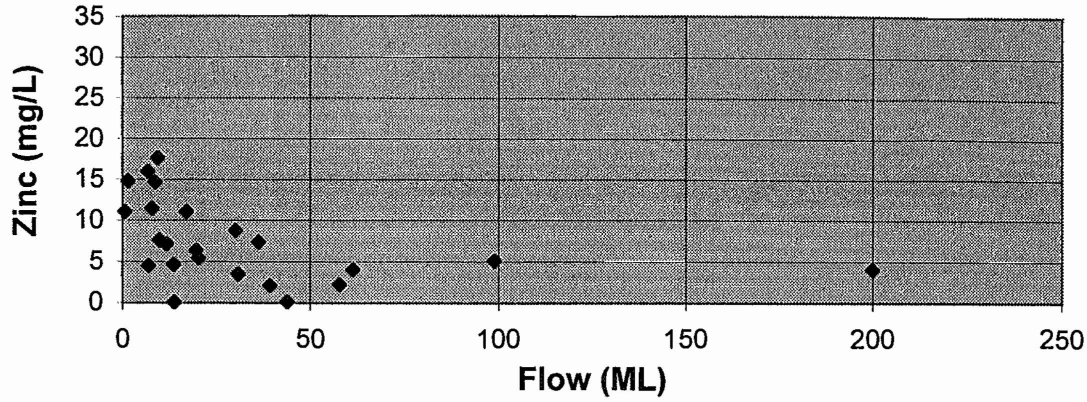


Figure 13
1998 Plot of Zinc (mg/L) versus Sulfate (mg/L)

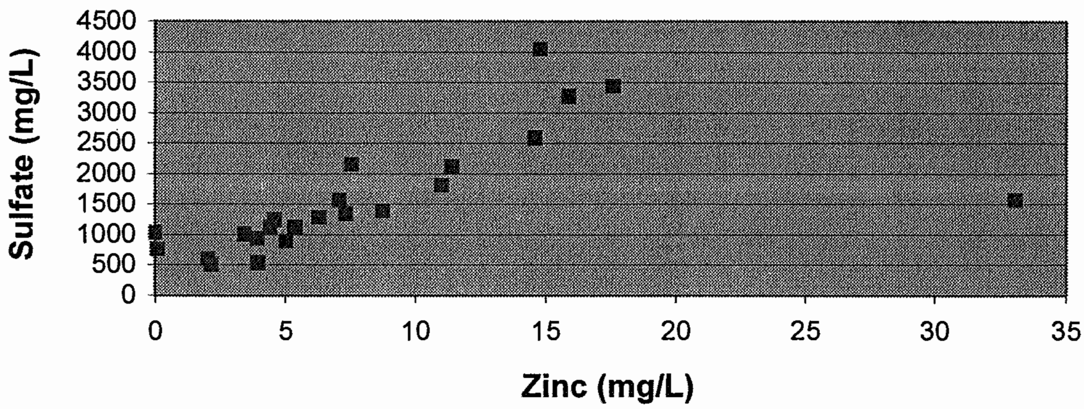


Figure 14
1998 Plot of Flow (ML) versus Cadmium (mg/L)

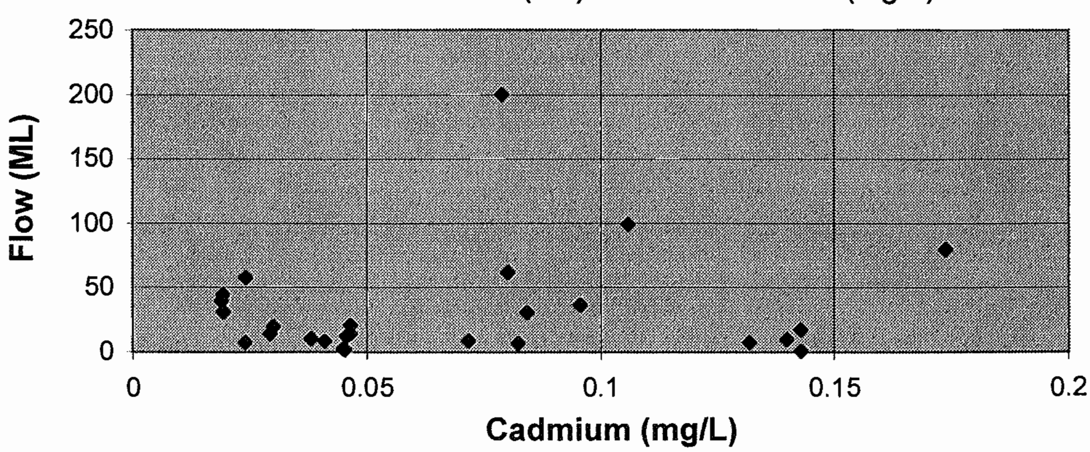


Figure 15
1998 Plot of Sulfate (mg/L) versus Cadmium (mg/L)

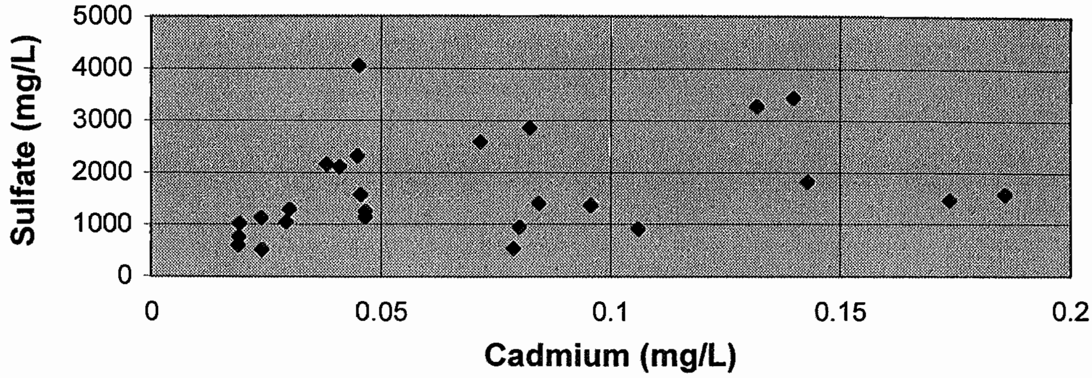


Figure 16
1998 Aluminium:Iron Ratio
(showing polynomial trendline)

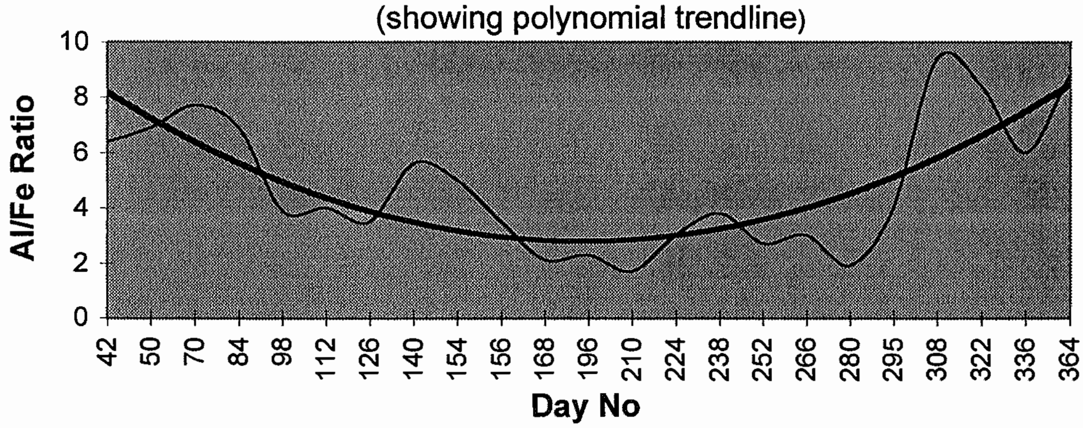


Figure 17
1998 Profiles of Scaled Cumulative Sulfate Load (kg/10)
and 'Total Iron' Load (kg)

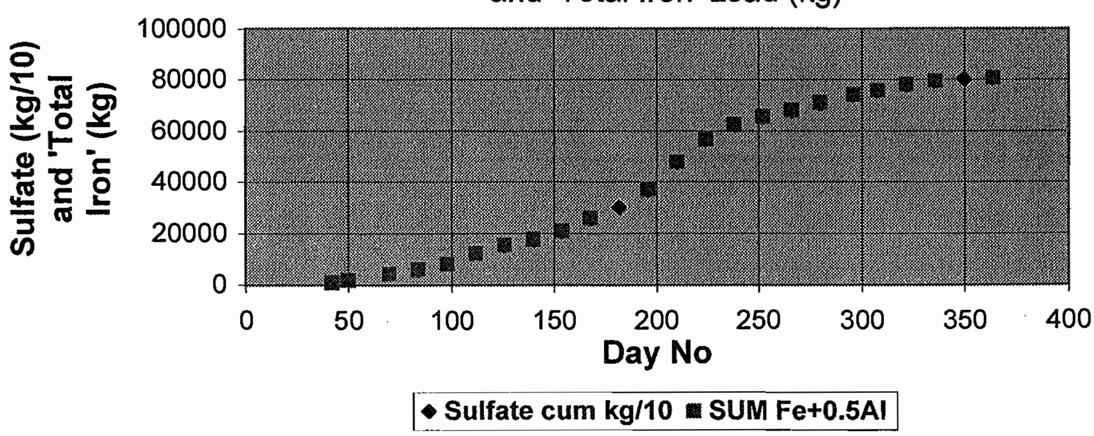


Figure 18
1998 Plot of Sulfate (mg/L) versus Aluminium (mg/L)

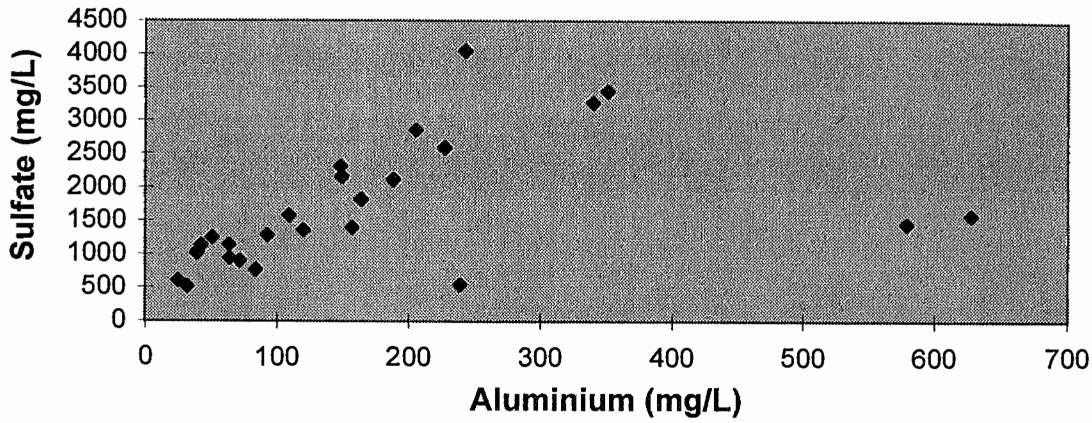


Figure 19
1998 Plot of Iron (mg/L) versus Sulfate (mg/L)

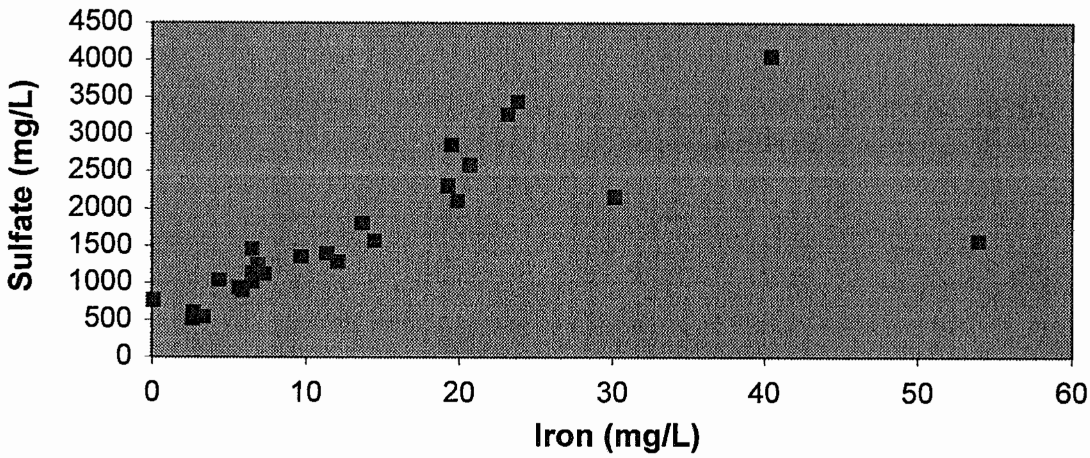


Figure 20
1998 Plot of Manganese (mg/L) versus Sulfate (mg/L)

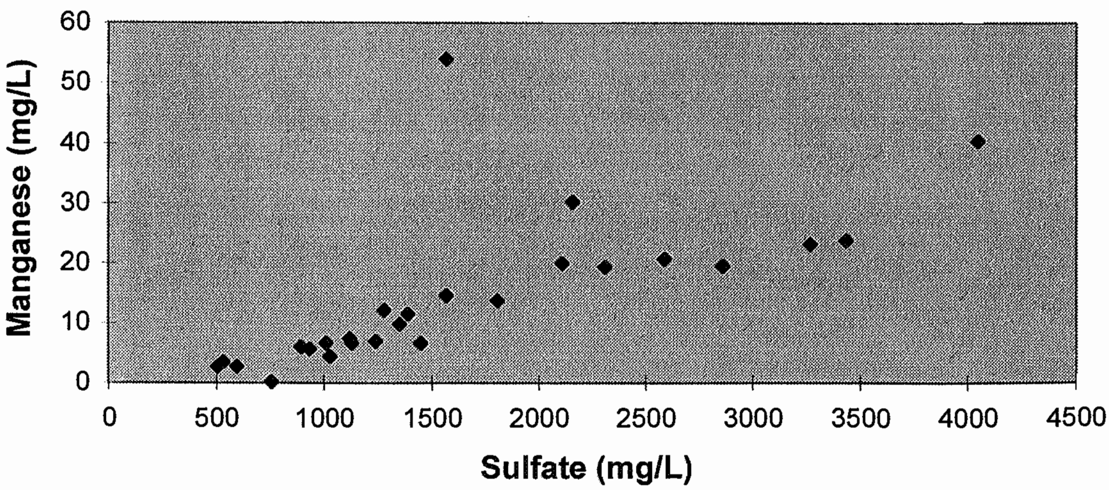


Figure 21
1998 Plot of Nickel (mg/L) versus Sulfate (mg/L)

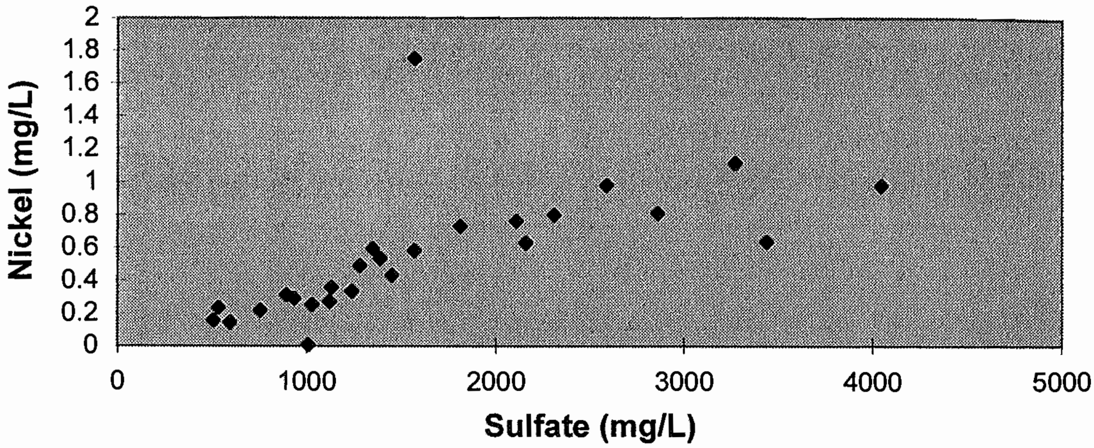


Figure 22
1998 Plot of Manganese (mg/L) versus Zinc (mg/L)

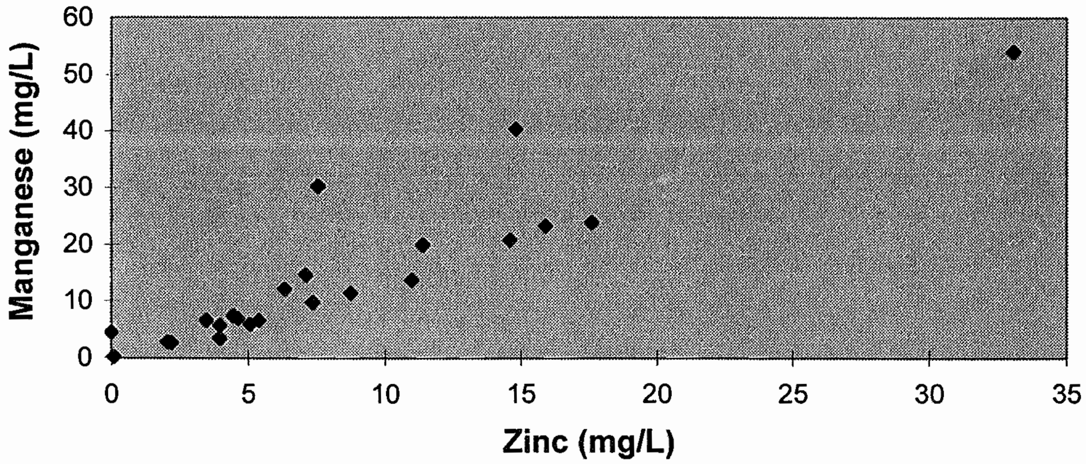


Figure 23
1998 Plot of Nickel (mg/L) against Manganese (mg/L)

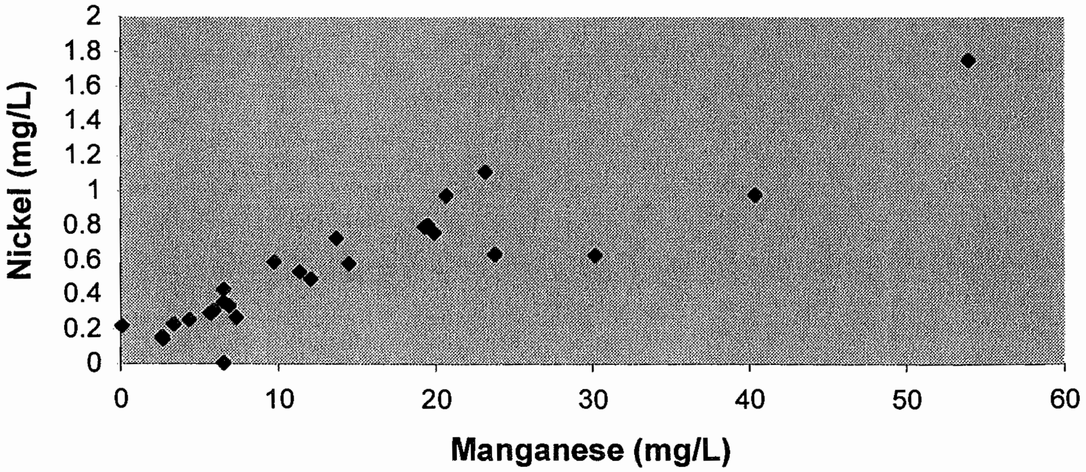


Figure 24
1998 Plot of Zinc (mg/L) versus Nickel (mg/L)

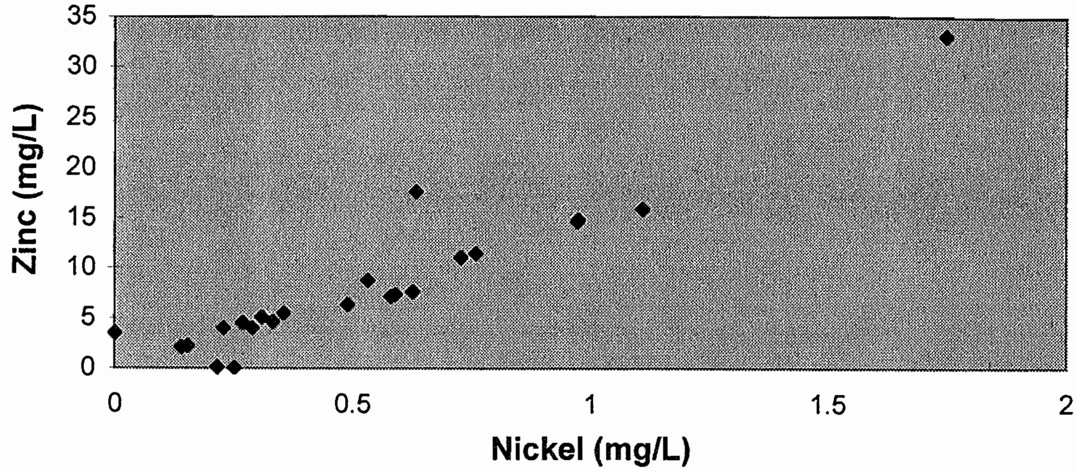


Figure 25
Scaled Metal Concentrations at Location 2. 1998

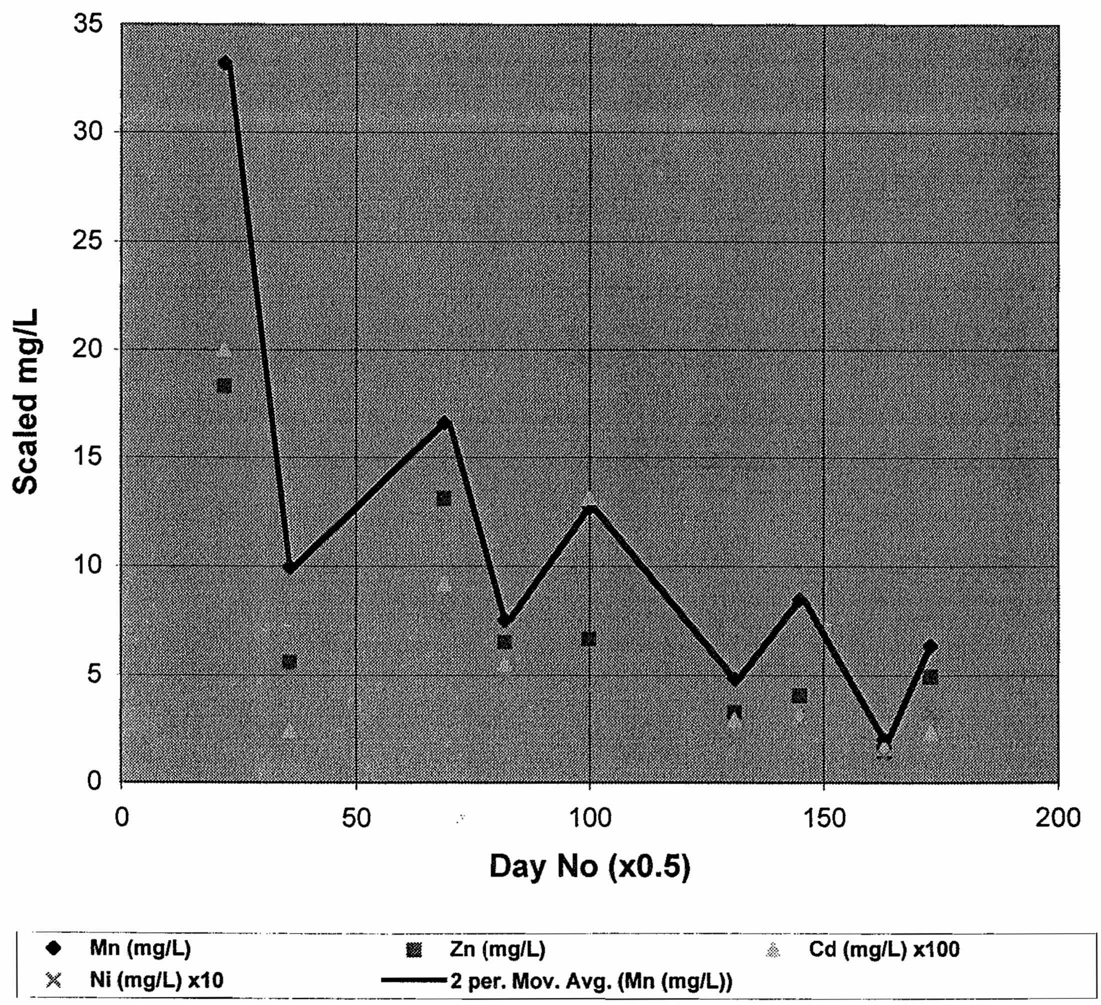


Figure 26
Scaled Metal Concentrations Location 5. 1998

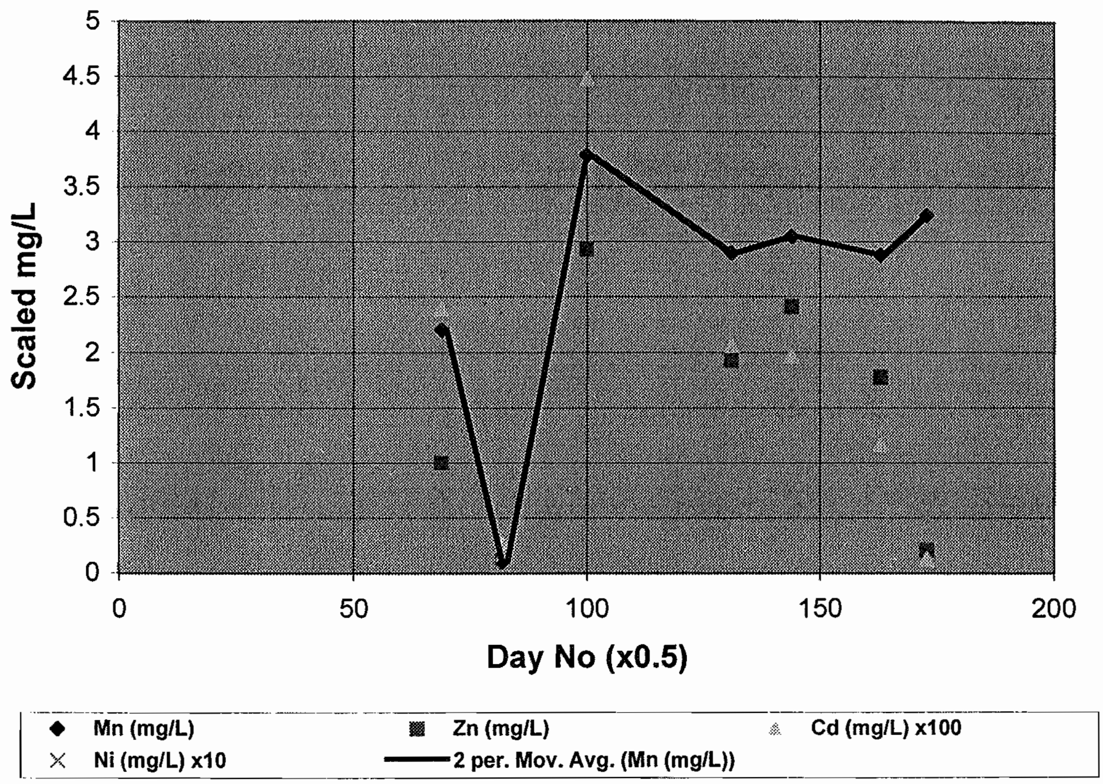


Figure 27
Scaled Metal Concentration Values Location 8

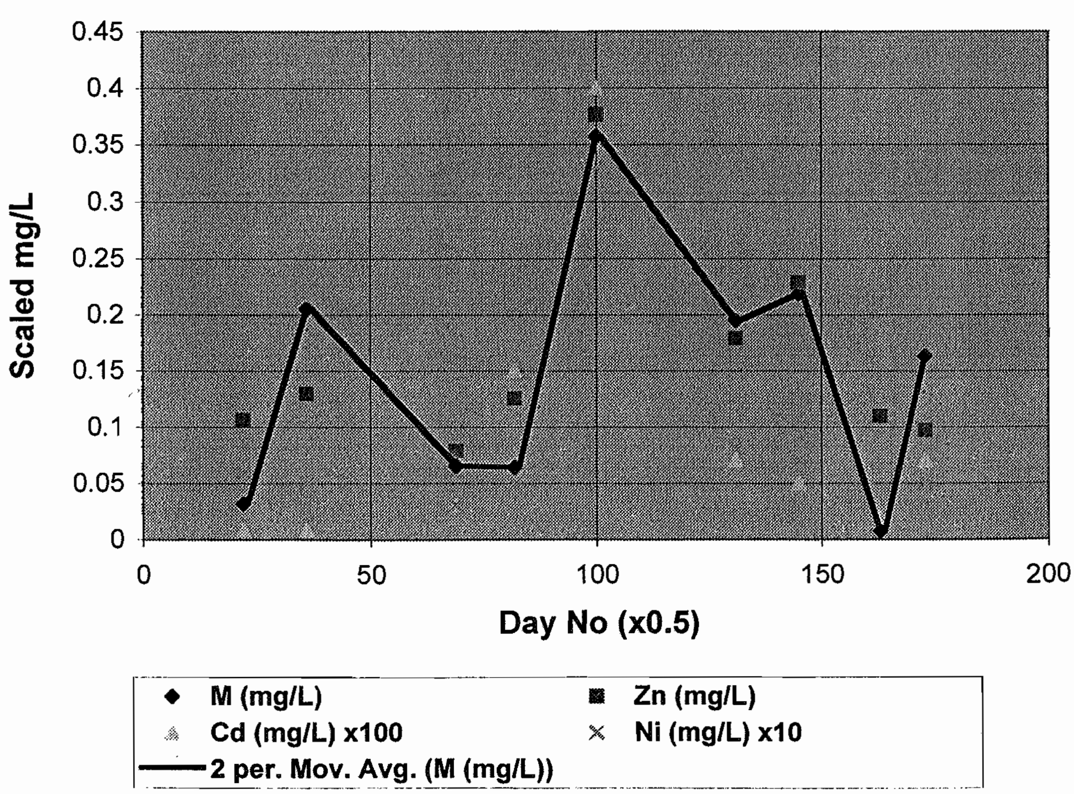


Figure 28
Cadmium Concentrations (x 100 mg/L) 1998

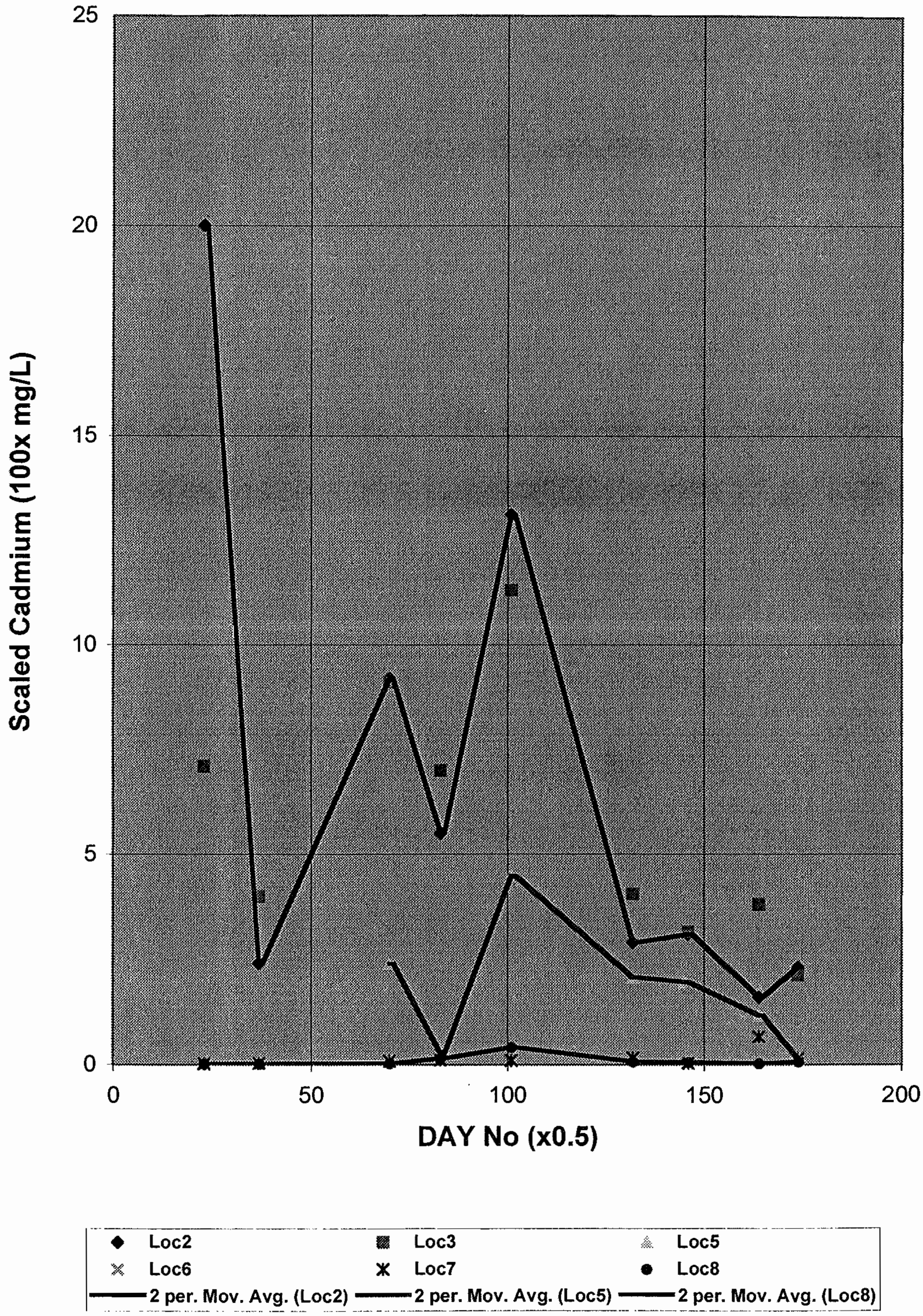
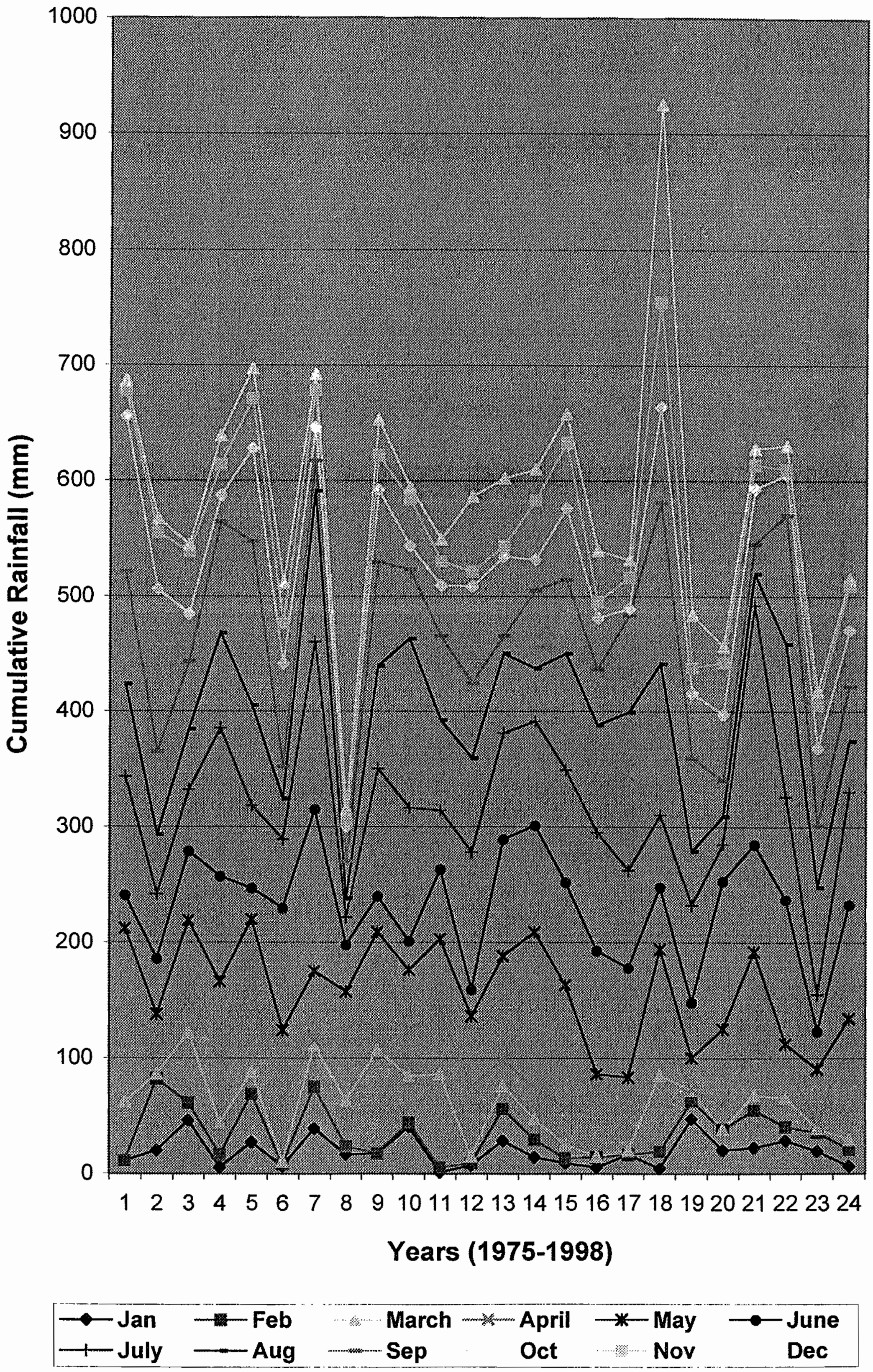


Figure 29
Cumulated Monthly Rainfall Profiles (1975-1998)



Graham Taylor – Mining Environmental Research, CSIRO[A]

Acid mine drainage at Brukunga – Impacts and remediation

Brukunga is an ideal laboratory for Honours students to study the impacts of acid drainage and potential remediation techniques. CSIRO Minesite Rehabilitation Research Program (now Mining Environmental Research) has sponsored a number of students, most of whom have not had the opportunity of reporting on their research. This paper is a synopsis of their work.

As a result of the oxidation of the tailings beneath the cover and vegetation, a variety of secondary minerals have formed, some of which are soluble and may contribute to the contaminant load of the leachate. Others (gypsum, ferrihydrite, jarosite, alunite and opaline silica) have formed a cemented layer 1.0–1.4 m below the tailings surface. Calculations by Agnew (1994) showed that this cemented layer had reduced the oxidation rate of the underlying tailings. However, there is still lateral movement of water below the cemented layer, adding to the acid drainage from the toe of the dam. Following this study, Agnew (1998) completed a PhD on the formation of hardpans/cements in tailings storage facilities throughout Australia, their stability and reduction in oxidation rate.

Organic ameliorants are commonly applied to acid-generating mine wastes to reduce oxidation and to supply nutrients for vegetative growth. Sewage sludge has been applied at various sites at Brukunga and is presently the subject of a PIRSA-sponsored investigation. Girdham (1994) studied:

- the effect of particle size on acid generation;
- the effect of sewage sludge on acid generation through exclusion of oxygen and formation of organic complexing agents; and
- the effect of sludge:rock ratio on generation of acid.

She found that rocks in the 1.0–3.0 mm size fraction generated the most acid through a moderately high surface area and sufficient pore size to allow penetration of oxygen. Using this size fraction, it was found that covering with sewage sludge reduced the acidity of leachate by two orders of magnitude together with a reduction in solute levels as a result of complexation. However, it was not possible to assess fully the effect of sludge:rock ratios on acid generation. Unfortunately, this study did not look at release of heavy metals from the sludge, the subject of present concern.

Elliott (1995) and Wilde (1995) both looked at local microorganisms as a potential means of remediating acid seepage. Elliott (1995) isolated and characterised acid-tolerant sulfate reducing bacteria and used cultures in a continuous flow reactor. Wilde (1995) isolated and characterised acidophilic algae and examined the feasibility of using them to remediate acid drainage in a rotating biological contactor. Whilst the sulfate reducing bacteria increased the pH, the algae werefound to accumulate some contaminant elements. Iron and Al were generally bioaccumulated, but other elements such as Mn and Zn were more species specific. What is the possibility of using a constructed wetland as a means of cleaning the water in Dawesley Creek?

Jia (1994) compared environmental management at Brukunga and Ranger. It is not surprising that in 1994 he concluded that the management program was unsatisfactory, the major limitation being lack of sufficient funds as well as knowledge. How far have we progressed in the past five years? Since the transfer of management from SA Water to PIRSA there has been an increase in funding (albeit not large) to gain more knowledge about the site and remediation techniques. In the future, it is intended to put an integrated technically sound proposal to Government for funding to minimise Brukunga Minesite issues.

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[A] A summary of work by M. Agnew, P. Elliott, J. Girdham, H. Jia and K. Wilde

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Measurement of impact of acid rock drainage on microbial communities using community level physiological profiling

Microorganisms perform vital biogeochemical functions in the carbon, nitrogen and phosphorous cycles in aquatic ecosystems. The measurement of their abundance, diversity and activity is an important component of measuring ecosystem health. Investigations aimed at the development of microbial measures as bioindicators of the impact of acid rock drainage (ARD) on aquatic ecosystems were initiated as part of the Managing Mine Wastes Project (MMWP) at ANSTO. The functional activity of microbial communities can be assayed by measuring community level physiological profiles (CLPP's) using BIOLOG microwell plates. This was one of a suite of techniques evaluated.

BIOLOG plates are commercially available 96 well microtitre plates where each well contains a freeze-dried carbon substrate (except controls), nutrients and a redox dye. Utilisation of the substrate is indicated by colour development which can be quantified by measurement of absorbance at 590nm. BIOLOG plates were originally designed for bacterial identification. Garland and Mills (1991) demonstrated the potential application of the plates for rapid characterisation of heterotrophic microbial communities from different habitats. BIOLOG plates have since been utilised in substrate utilisation profiling to characterise microbial communities in a range of environmental samples, including plant soil communities, soils, freshwater and groundwater. Microbial communities are characterised on the basis of the pattern and extent of utilisation of the carbon sources present in the BIOLOG plate selected for use. The pattern of microbial activities obtained on each plate gives a type of 'metabolic profile', representing functional or physiological attributes of the microbial community. Three types of BIOLOG plates were utilised in our investigations, the GN plate, designed for measuring activity of Gram Negative bacteria, YT plate for yeast activity and the ECO plate for measuring microbial activity of environmental samples (terrestrial and aquatic).

Sediment samples were collected from sites in the Dawesley Creek–Bremer River catchment area associated with the abandoned Brukunga pyrite mine on three separate occasions–November 1998, February 1999 and March 1999. A general microbial inoculum preparation method was followed which involved incubation of sediment with sodium pyrophosphate solution to detach microbes and low speed centrifugation to remove large particulates. The cell suspension obtained was then diluted before inoculation into the BIOLOG plates. The plates were incubated and absorbance values at 590 nm obtained using a microplate reader at selected time intervals. Absorbance measurement results were used to calculate an average well colour development (AWCD) value for each plate at a particular incubation time. an

Average well colour development is a measure of average colour development in a BIOLOG plate and is calculated by finding the average of all absorbance values for all relevant wells on a plate. GN and YT plates were used in November 1998, GN and YT in February 1999 and GN and ECO in March 1999. It is assumed to represent a measure of activity across the broad range of substrates in the plates and that it reflects both abundance and diversity.

The impacted sites showed low activity (lower carbon utilisation) on all plate types and at each of the three sampling times. Increases in metal and sulfate concentrations (from water quality analysis) correlated with decreases in AWCD values. Low AWCD values correlated with low pH at the acidic sites. Generally, an approximately ten-fold difference was observed in AWCD value between the most impacted ARD site and reference sites.

In November 1998, three sites were chosen to represent a site upstream of the ARD pollution source (3), an ARD impacted (4B) and a reference site downstream (9). Figure 1 shows the AWCD values of the assays for both GN and YT plates and clearly distinguishes between the ARD samples and the reference sites. GN and YT plates were consistent in distinguishing between impacted and reference sites.

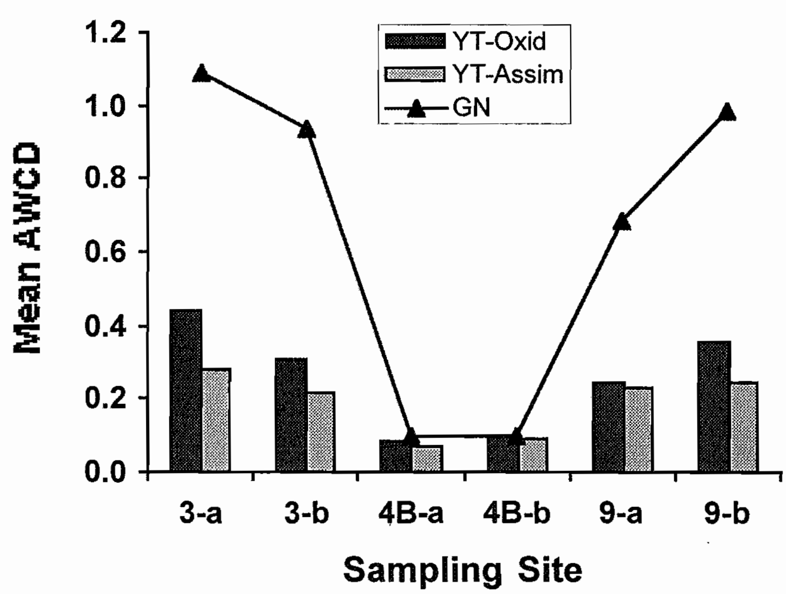


Figure 1. Mean AWCD values of Brukunga sediment data at 48 h for GN and YT microplates (November 1998).

Water samples from sites 4B and 9 were inoculated into duplicate GN plates after dilution. The AWCD values obtained were 0.15 and 0.9 respectively, indicating that similar values are obtained in the BIOLOG assay using sediment and water samples.

A more comprehensive set of eleven sediment samples was taken from Brukunga in February 1999. Figure 2 displays the AWCD values (mean values of triplicate plates) for both GN and YT plates. The acidic sites (4, 7 and 8u/s) had much lower AWCD values compared to the other sites ($P \leq 0.05$), with the exception of site 18 where lower activity was attributed to higher salinity levels. Analysis of variance results for 48 h GN data found there were significant ($P < 0.001$) differences in AWCD across sampling sites. The standard errors of differences for each sample site were similar (Harch 1999). It was observed that the same trend was evident for both GN and YT plates. The most severely impacted site (4) clearly demonstrated the lowest utilisation of carbon substrates in both BIOLOG plate types. A trend of increasing microbial activity was evident down the gradient of ARD pollution between sites 4 and 8 (4, 7, 8u/s and 8) which are situated on Dawesley Creek. Figure 3 highlights the relationship between some selected water quality parameters and GN AWCD values. The decrease in AWCD values was observed concurrently with an increase in metals and sulfate and a decrease in pH. In redundancy analysis for all 11 sites at 48 h incubation (GN data), it was found that the association between carbon source utilisation and levels of sulfate and pH were highly significant ($P = 0.005$ and $P = 0.015$, respectively). This indicates that levels of sulfate and pH were significantly impacting on carbon source utilisation.

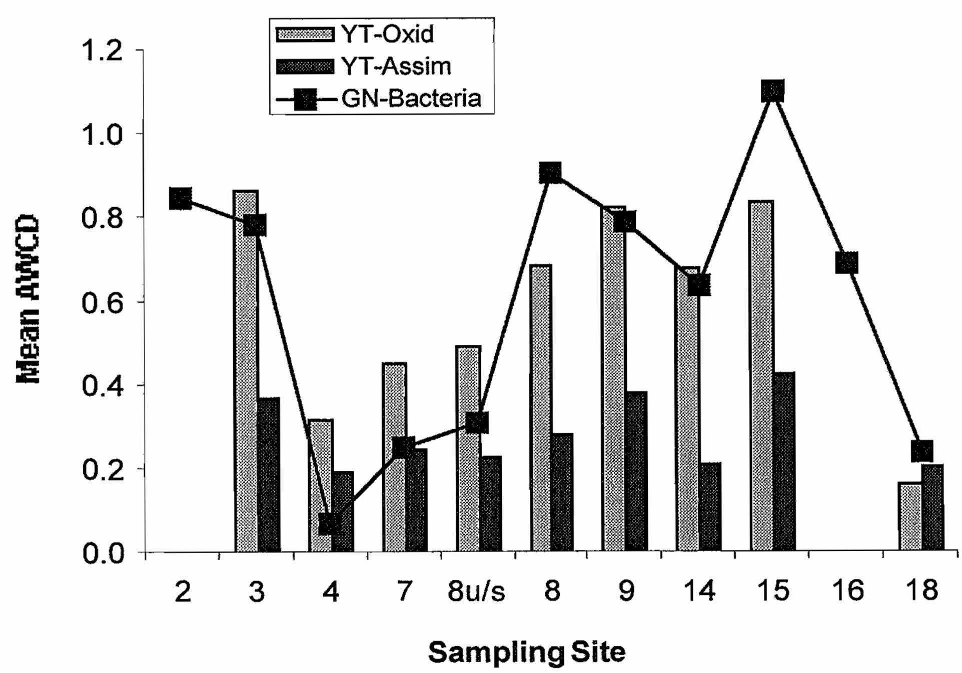


Figure 2. Mean AWCD values plotted by sample site for bacterial activity (48 h GN microplate data) and yeast activity (72 h YT microplate data) for sediment.

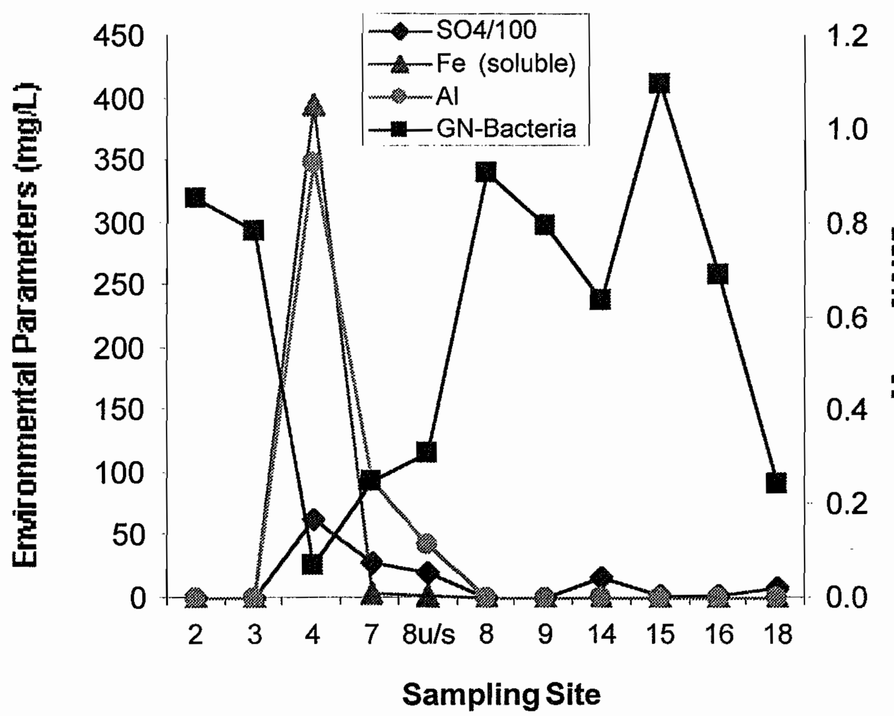
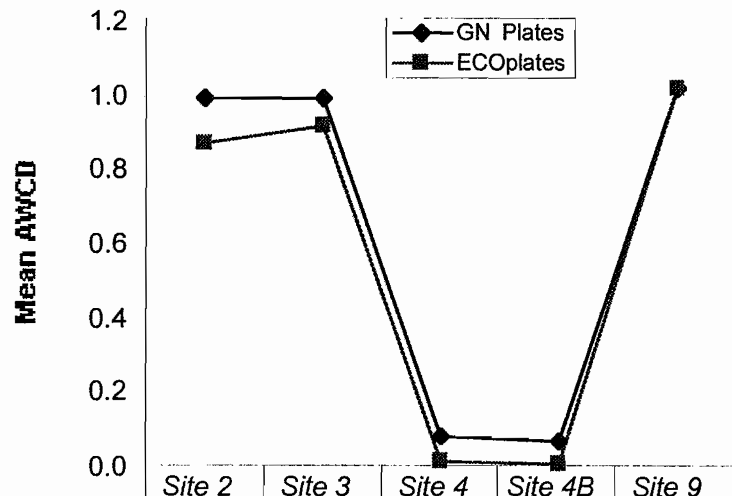


Figure 3. Profiles of selected water quality parameters and mean AWCD values for Brukunga sediment bacterial activity (GN microplates). Note that sulfate values have been scaled.

Results from the March Brukunga data confirmed the earlier observationsthat there was a clear distinction between the ARD impacted and reference sites. Both GN and ECO plates were examined at sites 2 and 3 (upstream of ARD impact), 4 and 4B (ARD impacted) and 9 (reference). ECO plates assay for bacterial activity but have three sets of a reduced set of 31 substrates on each plate (compared to 95 for GN plates). The comparison of results for both plate types was encouraging with very similar results obtained (Figure 4). The utilisation of ECO plates would enable a greater number of samples to be processed at one time as the need for triplicate and duplicate plate inoculations (as for GN) is eliminated.

Statistical analysis (univariate and multivariate) of the February GN data was performed by Dr B. Harch (Harch, 1999). Multivariate analysis revealed the relationships between the measured environmental data and microbial activity response on the BIOLOG plates. ARD, salinity and reference (largely nutrient impacted) sites were found to be distinguishable. A principal component analysis (PCA) of 48 h incubation data (AWCD data for replicate BIOLOG plates) for all eleven sites assayed is shown in Figure 5. In the analysis, anomalous replicates for sites 8u/s and 7 were removed. It was found that two components accounted for 84% of total variation as shown in the figure. Sites 4, 7, 8u/s (represented by 8*) and 18 which represent the most impacted sites (ARD and salinity), differ from all other sites along PC1. Site 8u/s differs from all sites along PC2.



	Site 2	Site 3	Site 4	Site 4B	Site 9
GN Plates	0.990	0.990	0.081	0.066	1.015
ECOplates	0.869	0.915	0.013	0.003	1.018

Sampling Site

Figure 4. Mean AWCD values for sites 2, 3, 4, 4B and 9 at 48 h incubation for both BIOLOG GN and ECO microplates (March 1999).

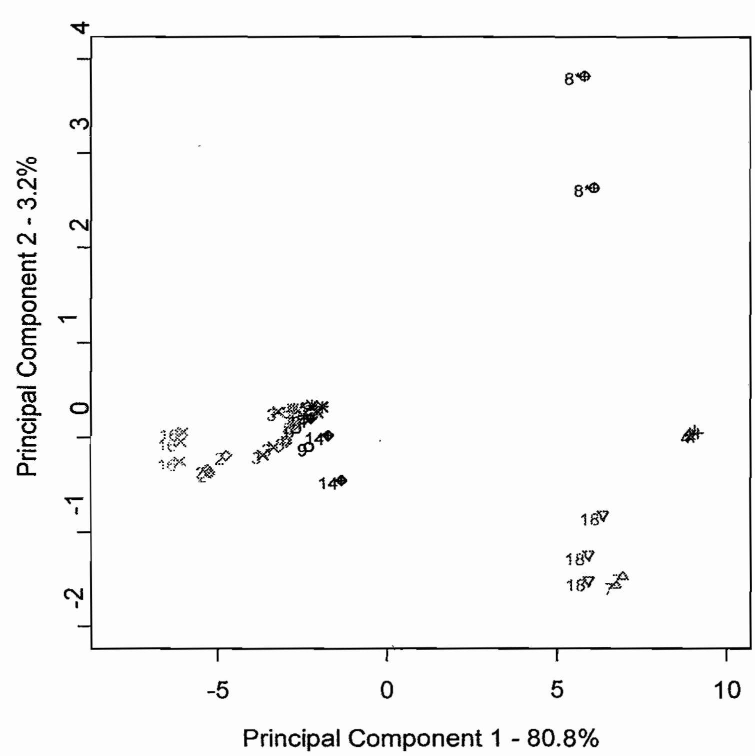


Figure 5. Plot of principal components 1 and 2 for analysis of BIOLOG GN replicate AWCD data for 11 sites at 48 h incubation.

Redundancy analysis of replicate-averaged 48 h data from all eleven sites in the February data set found that sulphate, chloride and pH were the best combination of environmental variables for explaining variation in the BIOLOG data. This is shown in Figure 6 where the utilisation of carbon sources increases with lower levels of sulphate along component axis 1. The carbon source arrows which form small angles with the pH vector, indicate the increased utilisation of these carbon sources as the pH increases (Harch 1999).

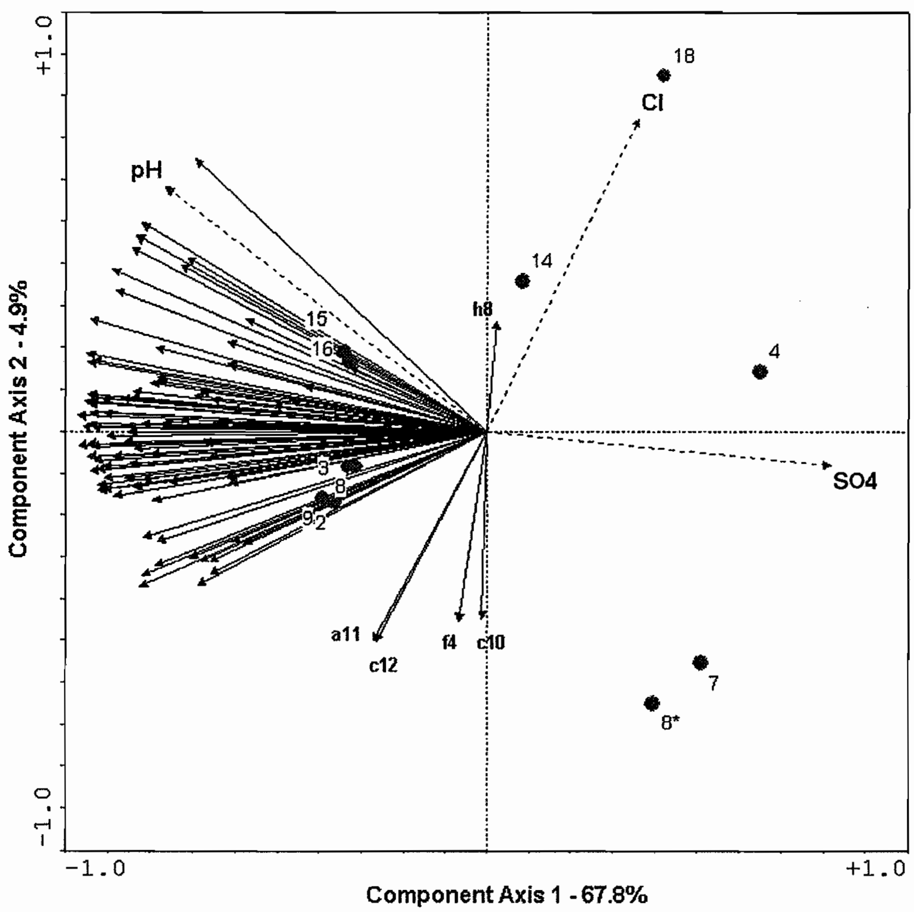


Figure 6. A redundancy analysis tri-plot of sample sites (dots), carbon sources (solid line vectors) and environmental variables (blue dashed vectors) based on the first two axes, with respect to sulphate, chloride and pH. The analysis was for 48 h replicate-averaged BIOLOG data.

Conclusions

The BIOLOG assay distinguishes between ARD polluted, salinity affected and reference sites suffering varying degrees of eutrophication. Carbon utilisation was significantly reduced at the ARD impacted sites (as indicated by low AWCD values). This result is for neutral pH conditions under which the assay was performed. It does not preclude the possibility acidophilic and/or other bacteria (e.g. anaerobic) are present and/or active at the ARD impacted sites. The results of the assays do however emphasise that the microbial population active at the reference sites against the substrates under the assay conditions used, is not present at the ARD impacted sites. The BIOLOG assay makes the assumption that the inoculum is representative of the diversity present in the microbial community and the assay only measures the fraction of the community capable of growth in the BIOLOG plates (a culture or growth dependent technique).

The results obtained were reproducible between sampling field trips and similar results were obtained for both sediment and water sample assays. Multivariate statistical analysis revealed the relationships between environmental parameters and microbial activity. ARD and salinity impacted sites were distinguishable. The utilisation of BIOLOG as a bioindicator of acid drainage provides data that is reproducible and ideal for multivariate statistical analysis.

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Phospholipid fatty acid analysis as a measure of impact of acid rock drainage on microbial communities in sediment: Comparison with other measures

Biological measures assessing the impact of pollution on aquatic ecosystems have been increasingly used over the last ten years to examine ecosystem health. The focus, however, has been on diversity and abundance of higher organisms, such as fish, frogs and macroinvertebrates, and it is desirable that such measures be made across all trophic levels of the ecosystem. In this study, the use of phospholipid-fatty acid (PLFA) analysis to quantify the effect of acid rock drainage (ARD), containing heavy metals and sulfuric acid, on the diversity and abundance of sediment microorganisms was evaluated and compared with other microbial measures of impact. Phospholipid-fatty acids are components of cell membranes and walls that occur in all microbes and their abundance and diversity in samples reflects differences in microbial community structure and abundance. In addition, PLFAs rapidly breakdown after death of microorganisms and thus total PLFA is a good measure of the total amount of viable microbial biomass present in a sample.

Acid rock drainage from the abandoned Brukunga mine in the Adelaide Hills, South Australia, has severely impacted on the Dawesley Creek/Mount Barker Creek/Bremer River system over the last 25 years. In Dawesley Creek immediately opposite the mine, water quality varies seasonally but is always acidic (pH 2.8–3.5). Significant pollutants include sulfate (1500–6330 mg/L), Zn (7.43–38.6 mg/L), Ni (0.44–2.29 mg/L), Mn (8.6–57.2 mg/L) and Al (109–348 mg/L). Other insults to the system include discharge of sewage treatment effluent into Dawesley Creek above the mine, agricultural and rural/urban run-off below the mine, and dry-land salinity affecting the lower reaches of Mount Barker Creek and the Bremer River. Sediment samples were collected from impacted and reference sites between November 1998 and September 1999 and subjected to a range of tests of microbial diversity, abundance, and activity. These included PLFA analysis, exoenzyme assays, community level physiological profiling of bacteria and yeast using Biolog (discussed in Wilde *et al.* 2000), Dimethyl sulfoxide (DMSO) reductase assays (a general measure of microbial respiration) and enumeration of microbes using conventional agar plating media.

Phospholipids from the sediments were solvent extracted quantitatively by the modified one-phase Bligh-Dyer method and purified by silicic acid column chromatography. Fatty acids derived from the phospholipid fraction were converted to fatty acid methyl esters (FAMES). These were identified and quantified using a gas chromatograph-mass spectrometer.

Fatty acids (26 in total) in the range C12–C20, consisting of saturated, terminally branched, hydroxylated, monounsaturated and polyunsaturated fatty acids, were detected in samples collected in November 1998. The total mass of PLFAs observed in a sample is a good measure of total viable microbial biomass and was clearly lower in sites affected by ARD as indicated by acidic pH and elevated metal concentrations in water samples. Sediment taken from (i) opposite the waste rock dump, (ii) the gauging station below the mine, (iii) Dawes Bridge and (iv) Balyarta Ford, all had far lower total PLFA values (0.45–0.95 mg/kg) than sediment from the reference sites, Peggy Buxton gauging station and Nairne Creek (2.8–4.7 mg/kg), or from further downstream in Mount Barker Creek and Bremer River (1.5–2.0 mg/kg). The sediments from ARD affected sites were also characterised by a higher proportion of polyunsaturated and hydroxylated FAMES when compared with reference sites. This was indicative of differences in microbial community structure.

Viable counts of microbes culturable at neutral pH on agar plates were conducted and a comparison with cell number estimates calculated using total PLFA showed similar trends, i.e. low values at ARD affected sites and higher values at reference sites. The plate method produced estimates 100-fold lower than total PLFA at reference sites, which is consistent with reports in the literature that only 1% of the microbes in an environmental sample are culturable in the laboratory. However, the 1000-fold difference between culturable organisms observed at ARD contaminated sites using the plate method and PLFA-based cell estimates was an exaggeration due to the plate method's failure to quantify acid tolerant microbes and consequent underestimation of the biomass. Both methods led to the conclusion that the microbial community observable at the reference sites was diminished in abundance and diversity at the ARD affected sites.

Statistical analysis using principal component analysis (PCA) of the variation of individual PLFAs and water quality data across sites produced a distinct cluster containing the ARD affected sites reporting with sulfate and heavy metals and another, far removed, cluster containing the reference sites, most of the PLFAs and the nutrients, phosphate and nitrate. The salinity affected sites of Bremer River and Mount Barker Creek (just above the confluence with the Bremer River) reported between the two clusters. Canonical correspondence analysis (CCA), multivariate hierarchical agglomerative clustering and non-metric multi-dimensional scaling were also used to identify similarities and differences among microbial communities, as indicated by PLFA profiles of the November samples. All of the above methods grouped the sites in a similar manner into two distinct clusters which corresponded with impacted and reference sites. Non-parametric, multidimensional, multivariate analysis using Primer[R] (BIO-ENV) software identified sulfate, Ni and Mn as being the water quality parameters that best described ($r = 0.8$) variance in the PLFA profiles.

Phospholipid-fatty acid analysis of sediment collected in February 1999, during a dry period, presented a more complicated picture. However, the pattern of the Peggy Buxton site (above the mine) having a high total PLFA followed by a marked reduction in sediments at the mine and a progressive increase at sites downstream was again observed. Biomass, as estimated by PLFA was quite low at salinity affected sites such as the Bremer River. The ARD affected sites again had a higher proportion of polyunsaturated PLFAs than other sites, confirming observations with November's samples, but in contrast did not demonstrate a clear trend of elevated hydroxylated PLFAs. Polyunsaturated PLFAs appear to be a positive biomarker for the impact of ARD on microbial flora of sediments. Canonical correspondence analysis again resulted in a cluster of ARD affected sites distinct from reference and salinity affected sites. Phospholipid-fatty acid analysis proved to be a cost-effective, rapid method of gaining valuable, statistically meaningful data on impacted aquatic ecosystems.

Measurements of exoenzyme activity in samples were also conducted using samples collected in February, June and September, 1999. Exoenzymes are proteins excreted from, or closely associated with the outer membranes of, microorganisms and perform key roles in the breakdown and cycling of carbon compounds, such as cellulose and proteins, and nutrients such as phosphorous and nitrogen compounds. They are produced by bacteria, fungi and algae. The activity of glucosidases, phosphatases, and aminopeptidases were assayed by adding substrates labelled with the fluorochrome methylumbelliferone to sediment samples and measuring the increase in fluorescence as the fluorochrome is released from the substrate by the enzyme. With the first set of samples collected, in February 1999, phosphatase was the only one of the three enzymes which gave activities indicating a consistent difference between reference and ARD impacted sites with activities being much lower at the latter.

However, subsequent analysis of samples collected in June and September, 1999 did not repeat this trend with some ARD affected sites having extremely high activities. Glucosidase activities did not yield clearly interpretable trends across sites in the three sample periods. Aminopeptidase activity, however, clearly corresponded to exposure to ARD in the June and September samples. The differences in activities between reference sites and ARD exposed sites were greater in the September sample set compared with those observed in June samples. In September, reduced aminopeptidase activity was observed as far downstream as Balyarta Ford, below the confluence with Nairne Creek. A similar phenomenon was observed with PLFA analyses of November 1998 and February 1999 samples, with the latter showing recovery at Dawesley Vale above the confluence with the Nairne, and the November samples showing low total PLFAs at Balyarta Ford below the confluence. This may correspond to the effects of a spring flush where although the pH of waters are higher than earlier in the year, the effect of ARD extends further downstream. This accords with results of macroinvertebrate surveys reported by Shultz (2000).

One factor that complicated the assay procedures for phosphatase and glucosidase, and interpretation of the data collected, was that the pH at which optimal enzyme activity was observed differed between the ARD-exposed and reference sites. Glucosidase activity in sediments from the gauging station at the mine and from Dawes Bridge was optimal at pH 4, whereas the reference sites showed optimal activity between pH 5 and pH 6. A similar result was observed with phosphatase but not aminopeptidase. The apparent relationship between aminopeptidase activity and exposure to ARD, as determined from multiple sampling times, and the similarity of pH optima for activity across sites indicates that it is suitable for further development as a bioindicator of ARD impact.

A comparison of total PLFA, and average well colour development using Biolog^R GN plates, for the November 1998 sample set, and macroinvertebrate diversity (average of three years) data obtained from Shultz (2000) showed that they followed similar trends. Values were low in proximity to the mine with slow recovery downstream. Results obtained using Biolog are treated in depth by Wilde *et al.* (2000). A comparison of results obtained with the February 1999 samples using different measures showed that Biolog (bacteria and yeast), culturable bacteria yeast and fungi, total PLFA, phosphatase, and DMSO reduction (a general measure of microbial respiration) all produced similar trends of low activity adjacent to the mine and down stream as far as the confluence with Nairne Creek. Exoglucanase enzyme assays (which measures breakdown of cellulose type compounds) were also conducted at fixed pH and also gave a similar trend. Likewise, assays for aminopeptidase activity, DMSO reduction, and Biolog^R using samples collected in June 1999 gave similar trends.

The following conclusions were reached:

- PLFA, Biolog^R, DMSO reduction, plate counts, and aminopeptidase assays produced similar trends of low activity at ARD exposed sites when compared with reference sites and appeared to correlate with water quality.
- Statistical analysis of PLFA and Biolog^R data indicated significant correlation between water quality parameters and these measures and ARD exposed sites were readily distinguished from reference sites using PCA and CCA.
- PLFA and Biolog^R analysis of November 1998 samples produced similar results to those observed by Schulz (2000) using macroinvertebrate diversity measurements.
- To determine ecosystem functional changes, activity measurements should be compared at different pH levels.
- Assays at fixed pH indicated disappearance of microbial populations at ARD exposed sites that were present in reference sites.
- Relatively high proportions of unsaturated PLFAs, and low pH optima for glucosidase activity, were indicators for ARD impacted sites.
- Microbial measures offer a relatively cheap and rapid method of quantifying impact of ARD on rivers and streams.

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Fungi at Brukunga: Community structure and metal tolerance

The aims of this study on the aquatic fungi of Dawesley Creek were encapsulated in three experiments, namely

1. To determine and record the fungal community structure at a site above and below the Brukunga mine;
2. To determine the effect of mine pollution on a fungal community, and
3. To determine the effect of metal pollutants from the mine on fungal growth.

Fungi were isolated from Dawesley creek in the Adelaide Hills. The sites used were DC 0.0, which is situated at the flow gate at Peggy Buxton road upstream of the Brukunga mine, and DC 2.1, which is at the flow gate 2.1 km downstream of the mine. Each site was sampled at the end of winter (September 1998) and the end of summer (March 1999).

The fungi were grown on substrate submerged in Dawesley Creek. The substrate consisted of leaves, twigs and bark that appeared to have come from river red gums (*Eucalyptus camaldulensis*). The substrate was brought back to the laboratory and broken into fragments of 1.0–1.5 cm^2. The fragments were placed on site water agar (plain agar prepared with water from the site from which the substrate was collected) and fungal isolates observed growing from the substrate were transferred to nutrient agar. For fungal isolates sampled downstream of the mine, the nutrient agar was modified using lactic acid to obtain a pH of 3.5.

Fungal community at each site

The fungal isolates were identified to genus level by microscopic examination of the fruiting bodies. Simpson's Diversity index was used to estimate the fungal community diversity and Jaccard's Similarity index was used to compare the taxonomic similarity of pairs of sites/samples. It was found that the diversity varied between sites and seasons. Diversity was not found to be different between the two sites, however, long term data are required for significant differences in diversity to be established. The similarity was greater for the same site between seasons, than for different sites within the same season. That is, the fungal community differed greatly between sites and less between seasons within a site.

Flow experiment

This experiment looked at the effect of polluted mine water on the structure of the fungal community from above the mine. There were four treatments:

1. Four weeks exposure to water from above the mine;
2. Two weeks exposure to water from above the mine followed by two weeks exposure to water from below the mine;

3. Two weeks exposure to water from below the mine followed by two weeks exposure to water from above the mine; and
4. Four weeks exposure to water from below the mine.

The experiment was repeated with water during summer and winter to determine if there were any seasonal differences. The results indicated that there was a definite shift in the community structure when exposed to contaminated water. There was also a shift in the community when the pollutant was removed, suggesting that the community has the ability to recover from a pollution event.

Metal tolerance:
The aim of this work was to determine the effect of eight metal pollutants (Al, Cd, Cu, Fe, Mn, Ni, Pb and Zn) from the mine on the growth of 12 fungal isolates which were selected to represent the fungal taxa collected at each site in each season. These isolates were grown in control or metal contaminated medium (up to the maximum metal concentration observed in recent monitoring) for five days and growth was assessed in terms of fluorescence released by fungal enzymes degrading Fluorescein diacetate. The results have been analysed using linear regression techniques only, and are summarised Table 1.

Table 1. Summary of linear regression analyses for a variety of taxon exposed to metals. ns, non-significant ($P > 0.05$); S, significant ($P \leq 0.05$); * negative slope; ** positive slope.

Taxon	Metal (as SO_4)							
	Al	Cd	Cu	Fe	Mn	Ni	Pb	Zn
Cylindrocarpon S	ns	ns	S*	S*	S**	ns	S*	ns
Stylopage	ns	ns	ns	S*	ns	ns	S*	ns
Gliocephalotrichum	ns	ns	ns	ns	S**	ns	ns	ns
Trichoderma S	ns	ns	ns	S*	ns	ns	ns	S*
Penicillium	ns	ns	ns	ns	ns	S*	ns	ns
Lacellina	ns	ns	ns	S*	ns	S**	S*	ns
Trichoderma W0	S*	ns	ns	ns	ns	ns	ns	ns
Cylindrocarpon W	ns	ns	ns	ns	S**	S*	S*	ns
Diploccocum	S*	ns	ns	S*	ns	ns	S*	S*
Trichoderma W2	ns	S*	S**	S*	S**	ns	ns	S*
Penicillium W	ns	ns	ns	S*	ns	ns	ns	S*
Chloridium	ns	ns	ns	ns	ns	ns	ns	ns

The results from Table 1 show that *Chloridium* was the only isolated species from the two sites that did not show any significant ($P \leq 0.05$) linear relationships with any metals. In general, the fits varied in their slope between various species and metals, with no obvious trends evident.

General conclusions

The general conclusions for the study were:

1. Both sites support reasonably diverse fungal communities, but with different taxa;
2. The community structure change can be recreated in the laboratory, and the structure is capable of changing back after the pollution is ceased; and
3. The effect of metal pollutants is determined by the isolate rather than the site it was obtained from.

Further studies

This study has provided a starting point for research into the use of fungal isolates as indicators for acid rock drainage. The next step would be to look at the synergistic effects of these metals and of low pH on the fungal isolates. It would be interesting to take the result found at Brukunga and see if similar results can be obtained at to other sites contaminated by acid rock drainage. These areas are similar in the method of contamination although the exact metal contaminants and rate of contamination can vary depending on the surrounding environment and climate. In addition, sites in tropical areas might have different fungal taxa because of temperature and substrate differences, with the same pattern of community change.

Further research could also look at the change in community structure due to contaminated water more closely. This study has shown that there is a shift and also the possibility of a shift-back. This needs to be looked at in closer detail to see exactly when the shifts occurred and how rapid they are. It would also be beneficial to look at the effect on the community structure of the substrate from DC 2.1 when exposed to water from DC 0.0. There is also a considerable amount of research needed into accurate and decisive methods for measuring biomass and growth in both small samples and aquatic hyphomycetes. The subject of fungal identification is a difficult area. There are great many genera of hyphomycete fungi and identification even to genus can be difficult. Within some genera there are hundreds of species (e.g. *Aspergillus*, *Penicillium*) and good taxonomic treatments are few and far between. However, modern methods of sampling microbial communities based upon polymerase chain reaction studies on DNA from environmental samples could prove useful in developing rapid methods for sampling aquatic fungi.

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Benthic diatoms as indicators of water quality in the Dawesley-Bremer catchment: A system affected by acid mine drainage, sewage pollution and salt – phase II

This study utilises diatoms as biological indicators of water quality and describes the changes in species diversity and abundance in the Dawesley-Bremer catchment using multivariate statistical techniques from samples taken during a low flow period in March 1998. The project had the following aims:

- To examine the impact of acid mine drainage on the benthic diatom populations of the Dawesley-Bremer catchment.
- To examine the length of the pollution gradient downstream of the mine.
- To compare and contrast these findings with the December 1997 data.

Major findings:

- The contrasting nature of pollution in the Dawesley-Bremer catchment yields discreet diatom communities. This is more pronounced in the 1998 data.
- In 1997, the variation in diatom species was best described by:
  - high TP/high TKN
  - low pH/high metals
  - high conductivity
- In 1998, the variation in diatom species was best described by:
  - low pH
  - high conductivity
- The impact of the mine was not detectable below the confluence with Mount Barker Creek in 1997. In 1998, the impact of the mine was not detectable below the confluence with Nairne Creek.

There was pronounced co-variation of the environmental variables in the 1998 data and further statistical analyses may need to be undertaken.

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Algae at Brukunga: What do we know?

Introduction and Rationale

The Brukunga pyrite mine site is situated in the Mt Lofty Ranges near Adelaide (South Australia), in an area with predominantly winter rainfall and a long period of low rain and high evaporation that usually dries most local creeks and rivers into chains of ponds over summer and early autumn. The mine site lies in the catchment of Dawesley Creek, which flows via Mount Barker Creek into the Bremer River and to Lake Alexandrina. Acidic leachate from the sulfidic waste rock and tailings at Brukunga drains into Dawesley Creek.

ANSTO's Managing Mine Wastes Project (MMWP) aims to provide a sound scientific basis for decision making on management of mine wastes. Algal studies have formed part of the broader biological investigation of Brukunga, under the auspices of the MMWP. An important rationale for these studies was that biological response to acid mine drainage (AMD), rather than either the chemistry or the physics of generating AMD, is ultimately at issue and that measures of ecological detriment must include biological measures. Further, we believed that an ecosystem response would not be adequately characterised using only a single group from the many biotic groups that make up an aquatic ecosystem. This presentation summarises the findings of two studies of algal responses to AMD in Dawesley Creek. Some attempt is also made to place these studies in context by comparing them directly with one another, and also with the findings from studies of macro-invertebrates and fish in the same area.

Methods

Both algal studies used a standard set of ~20 sampling sites from Dawesley, Nairne and Mount Barker Creeks and a part of the Bremer River (Fig. 1). Sampling was confined to the dry period in summer (December 1997) and early autumn (March 1998), when the effect of the AMD was expected to be most apparent. In both studies, water was sampled for a variety of chemical (and some physical) analyses with which to compare the measured biological responses. Most chemical analyses were undertaken by the Australian Water Quality Centre (AWQC). Statistical techniques were used to compare chemical and biological information so as to reveal underlying patterns.

The diatom study (see Sincock *et al.* 1999a; Sincock *et al.* 1999b; Sincock 2000) relied on the community structure of benthic diatoms growing in the field as a biological measure of water quality conditions. Diatoms were sampled from both rocks and mud at the various sample sites, prepared, identified and counted (microscopically) to give the relative abundances (RA's) of some 70 species expressed as a percentage of the total number of diatoms at each site. A multivariate statistical analysis, canonical correspondence analysis (CCA), was used to compare the diatom and water quality data sets, after exclusion of RA's for 'rare' diatom species and water quality parameters for which the data were incomplete (usually because >50% of data fell below the detection limit). Ordination refers, collectively, to techniques that arrange sites along axes according to the species composition found at each site. The aim is to plot sites that have similar species composition close together and those with dissimilar species composition further apart (Jongman *et al.* 1995). Canonical ordination, such as CCA, detects patterns of variation in species composition data that can be 'best explained' by the observed environmental variables. The CANOCO program enabled us to correlate each environmental variable with the biologically determined ordination axes, revealing the most biologically important environmental gradients. CANOCO usually presents data for four ordination axes, of which only the first two are plotted here—the first axis always accounts for the greatest variation in the biological data with subsequent axes 'explaining' relatively less.

The laboratory-based fluoroscein diacetate (FDA) study (see Regel *et al.* 1999), in contrast, used a 'metabolic probe' in combination with flow cytometry to assess the short-term physiological responses of laboratory-grown algae exposed to water sampled from the field. A green and a blue-green alga were exposed to water from the sample area for 1 and 24 h periods. Enzyme activity (esterase) was measured by brief exposure to FDA, which fluoresces only after it has been metabolised by the alga. Fluorescence of individual algal cells was then measured in the flow cytometer. Data were expressed as mean rates of FDA conversion (esterase activity) relative to the rate achieved by 'control' algae grown in their ordinary WC medium. Also, the use of flow cytometry enabled us to define metabolic states, relative to the controls: S_1 = reduced activity, S_2 = normal (*i.e.* equivalent to control algae) and S_3 = stimulated. The experiment involved two different algae exposed for two periods and with two measures of the algal response.

Results

Water chemistry (multiple pollutants)

In summer, much of the flow in the upper Dawesley Creek (upstream of the Brukunga mine site; sites 1–3 in Fig. 1) comes from the Bird-in-Hand waste water treatment plant (BIHWWTP). This effluent is secondarily treated and, as such, still has very high concentrations of the plant nutrients, nitrogen and phosphorus. Total phosphorus (TP) and nitrogen (TN) exceeded the ANZECC 1992 guidelines for the preservation of aquatic ecosystems by more than a factor of 100 and 10, respectively (Table 1). The stream section downstream of the mine site to just upstream from the confluence of Dawesley and Nairne Creeks (sites 4–7 in Fig. 1) was worst affected by AMD, with pH as little as half the minimum recommended by ANZECC (1992) (Table 1) while aluminium concentrations were more than 100 times the ANZECC recommended maximum. Ironically, the large amount of phosphorus found upstream of this section was precipitated as metal phosphates by the AMD (effectively a tertiary treatment for phosphorus) such that TP fell to quite low concentrations.

Downstream from Nairne Creek, but upstream of the point at which Dawesley Creek flows into Mount Barker Creek (sites 11–14 in Fig. 1), the relative contributions of Dawesley and Nairne Creeks changed the water quality from evidently AMD-affected in December 1997 to being essentially that of the nutrient enriched Nairne Creek in March 1998, making this an especially variable aquatic environment (Table 1). Mount Barker Creek accounted for most of the flow downstream from its confluence with Dawesley Creek. Mount Barker Creek (sites 16 and 17 in Fig. 1) also yielded the 'best' water quality we sampled, despite exceeding the ANZECC (1992) guidelines for conductivity and nutrients, particularly. The Bremer River water was quite saline (site 18 in Fig. 1; Table 1), especially after flow ceased and evaporation contributed to conductivities 5 to 10 times the maximum recommended by ANZECC (1992).

Table 1. Ranges for selected water chemistry variables in five stream sections in the sample area during 'summer'.

Site(s)	TP ($mg\ L^{-1}$)	TN ($mg\ L^{-1}$)	pH	Aluminium ($mg\ L^{-1}$)	Conductivity ($\mu S\ cm^{-1}$)
1–3	**16.8–26.4**	**9.5–52.8**	6.7–7.6	0.03–1.01	1600–2400
4–7	0.04–0.25	1.3–4.8	**2.7–3.2**	**81–239**	2000–6400
11–14	**0.06–7.5**	1.4–6.1	**4.3–7.5**	**0.05–27.3**	2200–10700
16–17	0.08–0.26	1.0–1.3	7.0–8.1	0.06–1.06	2900–6000
18	0.02–0.13	1.0–2.7	7.5–8.8	0.04–0.22	**8100–19700**

The study area has at least three significant sources of pollution (treated sewage, AMD and salinity) that tend to dominate in particular stream sections. However, in at least one stream section the water quality is affected by two of these pollutants and can vary considerably, depending on the flow. This 'AC/DC' section probably presents special challenges to the biota that live there.

Diatom communities

The aim of this study was to assess diatoms as indicators of water quality, specifically in relation to AMD. The study was field-based and involved taking very small samples from rocks and mud, according to an agreed Australian protocol for sampling benthic diatoms. A high level of taxonomic expertise was required to process these and it was this area that required the greatest effort. Statistical analysis of the data was also a relatively complex task. A realistic minimum timescale for completion of such a study is probably 2–3 weeks from the date of the field visit, although this could be reduced to 1–2 weeks by familiarity with the site and its diatom flora.

Figure 2 shows a CCA plot for selected diatom species (3 letter abbreviations of genus and species names) and an overlay-plot of the two samples taken from each site (*e.g.* 1l = rock-associated diatoms from site 1 and 1p = mud-associated diatoms from site 1). A group of four diatom species is clearly separate from the others along axis 1 (we were not able to identify 2 of these diatoms to species level) and this group was dominant at sites 4–7, the most AMD-affected sites. Site 8 was intermediate because the pool was so close to the junction with Nairne Creek as to be flushed with Nairne Creek water at times. Axis 1, the one explaining the greatest amount of the variation in the diatom assemblages, was best correlated with water quality parameters indicating AMD pollution (*e.g.* pH and aluminium). The bulk of recorded diatom species form a more or less continuous group, best aligned with axis 2. Sites 1–3 were the most nutrient-rich sites and they plot furthest up axis 2, while sites 13 and 14, with amongst the lowest concentrations of available phosphorus, are positioned at the bottom end of axis 2. Axis 2, explaining somewhat less of the biological variation, was best correlated with nutrient concentrations. However, nutrients and salinity tended to be negatively correlated (high nutrients associated with low salinity and vice versa) so that axis 2 also tends to isolate diatom species according to their preferred salinity. We can also identify axis 2 as related to both nutrients and salinity on the basis of our experience of national diatom floras as well as from international research into the ecological 'preferences' of particular diatom species. The finding of a diatom species that 'prefers' to grow in saline conditions in Europe or North America allows us to infer that 'preference' in Australia, for the same species. This is only possible because diatoms are generally considered to be cosmopolitan, unlike many higher organisms that have species or genera confined to particular regions, like Australia.

The analysis shows that AMD-affected sites were the most distinctive, in terms of their diatom flora. It could be asserted that AMD was the 'worst' pollutant in the system, reducing diatom community diversity and selecting for the few species of diatoms able to tolerate the combination of very acid and metal-polluted water. It is clear from the analysis that changes in the community structure of benthic diatoms were able to distinguish the effects of AMD from those of nutrient pollution and probably saline pollution as well.

Flouroscein diacetate and flow cytometry

The aim of this study was to develop and test a relatively new algal assay for water quality, again, focussing on the assessment of AMD. The study was primarily laboratory-based but involved sampling and transporting several litres of water from each field site. A significant part of the effort involved in this study was in laboratory preparation (cultures etc.) prior to the field visit. A realistic timescale for completion of such a study is probably two days after the field visit.

Figure 3 shows the esterase activity and the number of algal cells in the 'reduced activity' state, S_1, after a 24 h exposure to the field-sampled water (1 h data are not shown)—both measures are given as percentages of 'controls' (error bars are standard deviations of triplicated experiments). The two very different algae used in these experiments are labelled 'Sel.' for *Selenastrum capricornutum*, a green alga, and 'Mic.' for *Microcystis aeruginosa*, a blue-green alga or cyanoprokaryote. Esterase activity in *Selenastrum* (but not *Microcystis*) was greater than 150% of 'control' activity after 24 h in water from some nutrient-affected sites, indicating that the green alga was stimulated by exposure to excess nutrients, while the blue-green was not. The number of cells in S_1, a state indicating stress, was increased in water from the AMD-affected sites (4–7) and this corresponded to a reduction in the esterase activity, albeit less clearly. The best indication of the presence of AMD in the sampled water was the number of *Microcystis* cells in S_1. This is, perhaps, counter-intuitive (see Fig. 3) but is based on the somewhat better correlations found between the number of cells in S_1 and both pH and aluminium concentration for *Microcystis* compared with *Selenastrum*, and in 24 h compared with 1 h exposures. In 1 h exposures, both algae showed reduced esterase activity and a corresponding increase in cells in S_1, in water from site 18 (data not shown). Both algae had recovered from this 'saline shock' within 24 h (Fig. 3).

The experiments show that AMD was best indicated by the stress response of the blue-green alga (*Microcystis*) after a 24 h exposure to the field-collected water, although both algae were stressed by exposure to AMD. In all, the combination of two algae with two exposure times and two measures of physiological response was able to rapidly distinguish the three pollutants affecting the stream ecosystem being studied.

Discussion

Comparing the two studies - a practical view

We have tested two algal biomonitoring tools at Brukunga. Both were successful in showing the ecological impact of AMD and apparently distinguished between the major pollutants affecting Dawesley Creek and the other streams studied. In practical terms, the diatom study required more time and a more specific knowledge, that is, how to identify diatoms to species level. Nevertheless, the fact that the diatom study gave information about the biota actually growing in the streams of the study area provided independent 'field-truthing' for ecological risk assessment (see Twining 1999, 2000) and should also be directly useful in setting rehabilitation targets for Dawesley Creek. The FDA study provided a rapid algal bioassay of water quality using physiological 'stress' to measure the acute effects of exposure to three pollutants. The comparatively novel protocol was faster than the more widely used growth-based algal bioassays, and the use of a blue-green alga was apparently well suited to 'detection' of AMD.

Comparing the algal studies with studies of macro-invertebrates and fish

A question that arises from the rationale for the algal studies (see above) is whether or not different groups of organisms are giving the same picture of the ecological effects of AMD at Brukunga. In Fig. 4 we use the total number of taxa found at a given site as an indication of the diversity of a biological community.

Evidently, the diversity of diatoms and macro-invertebrates was reduced in the area worst-affected by AMD at the time of sampling (December 1997). What appears different is that a low diversity of macro-invertebrates persisted in Dawesley Creek downstream of its confluence with Nairne Creek, while for diatoms the diversity recovered at site 11 and remained at reference levels thereafter. In this direct comparison of data, the two biomonitors give a consistent picture of ecological detriment from AMD but with some difference in detail. The stream section including sites 11–14 has been identified as a section in which water chemistry can change according to the relative flows in Dawesley and Nairne Creeks (see *Water chemistry*, above). In the event of a recent improvement in water quality in this stream section, we speculate that the lack of recovery in diversity of the macro-invertebrate community represented a slower response timescale relative to that of the diatom community. That is, the diatoms were able, through their rapid growth rates, to exploit the improved water quality, while the macro-invertebrate community had yet to recover from the effects of more concentrated AMD. According to Hicks (1997) no fish were found in Dawesley Creek upstream from the point at which it flows into Mount Barker Creek, in a study carried out in 1997. This may result, in part, from the even slower response timescale that would be expected from the more complex life-cycles of fish compared with those of many macro-invertebrates. The timescale of variation in water quality may be too short for fish to establish and breed in this 'AC/DC' section of Dawesley Creek. Whatever the explanation, another point to note is that fish were simply absent from the most polluted stream reaches and, although this is to some extent ecologically informative, there is no further ability to distinguish either the types or extent of pollutants further upstream. In contrast, both diatoms and macro-invertebrates were present at the most AMD-affected sites and upstream in the nutrient-polluted stream section so that their communities were statistically analysable throughout the study area.

Concluding comments

The algal studies have taken a catchment-scale view of water quality and successfully demonstrated the use of two biomonitoring approaches in a system affected by several pollutants of economic significance to Australia. Both algal water quality measures seemed able to distinguish between the three pollutants (nutrients, acid mine drainage and salinity) that affect the Dawesley Creek-Bremer River system.

In the larger context of managing mine wastes in Australia, we are moving towards a greater role for the use of biological information to measure the ecological effects of pollutants such as AMD, rather than relying simply on physico-chemical water quality data. Further, we are developing a suite of biomonitoring tools from which to select the one(s) appropriate to particular situations, rather than attempting to define a 'panacea' group to be applied in all circumstances.

Acknowledgements
We thank Peter Schultz (AWQC) for access to his macro-invertebrate data and also the staff of the AWQC chemical laboratories for their help with the water chemistry.

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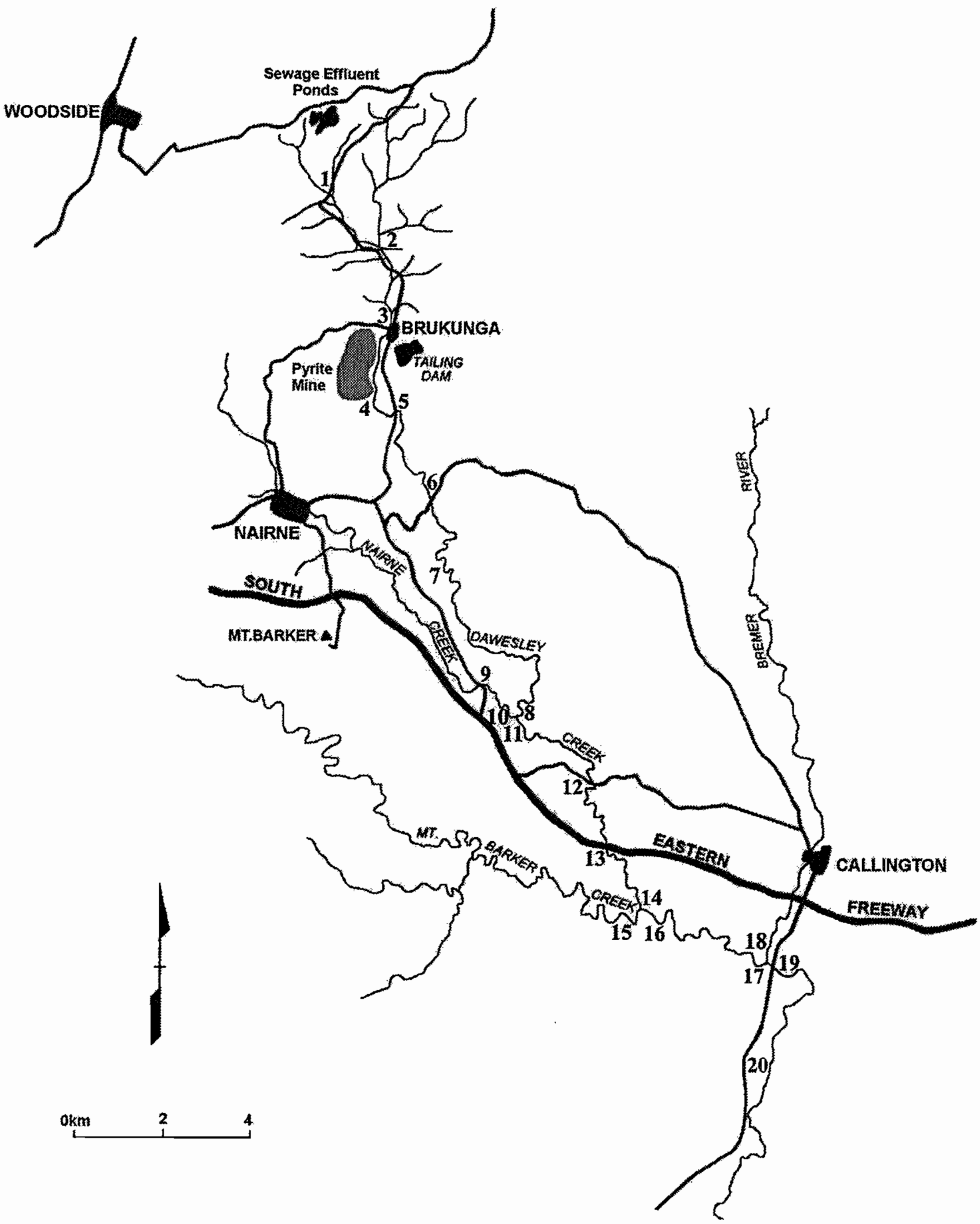
Figure Captions

Figure 1. The Brukunga study area. Sampling sites are indicated as numbers from 1–20.

Figure 2. Canonical correspondence analysis plot of species data (upper panel; see Sincock *et al.* 1999b for key to 3 letter abbreviations of genus and species names) and an overlay plot of the two samples taken from each site (lower panel; e.g. 1l = rock-associated diatoms from site 1 and 1p = mud-associated diatoms from site 1).

Figure 3. Esterase activity (upper panel) and the number of algal cells in the 'reduced activity' state, S_1 (lower panel), after a 24 h exposure to the field-sampled water. The x-axis gives distance downstream from the BIHWWTP and at the top of each plot is a line drawing that shows the position of inflows (N = Nairne Ck, MB = Mount Barker Ck and B = Bremer River). Site numbers are given just below this line, with closely adjacent sites in brackets. Reference sites, those unaffected by flow in Dawesley Creek, are grouped to the right of the vertical line and arbitrarily positioned along the x-axis. 'Sel.' = *Selenastrum capricornutum* and 'Mic.' = *Microcystis aeruginosa*.

Figure 4. A simple indicator of community diversity (*i.e.* the total number of taxa found at each site) for diatoms and macro-invertebrates, sampled within a day or so of each other, in December 1997. The x-axis, line drawing and position of reference sites are as for Fig. 3. Site numbers are plotted for the diversity of diatoms.


Sewage Effluent Ponds
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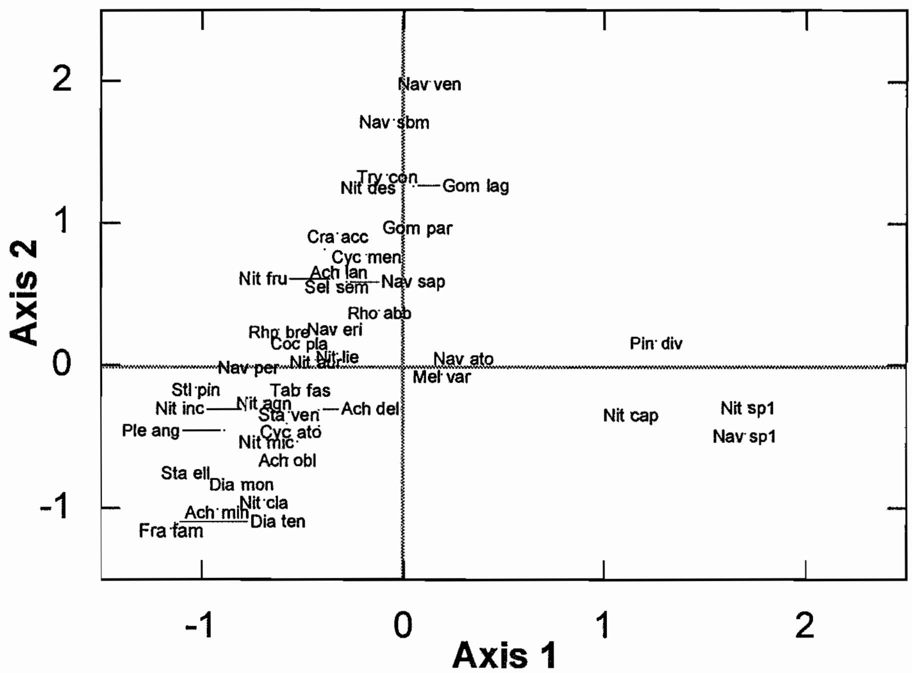
1
2
3
BRUKUNGA
TAILING DAM
Pyrite Mine
4
5
6
NAIRNE
NAIRNE CREEK
7
SOUTH
MT.BARKER
DAWESLEY
9
8
10
11
CREEK
12
MT. BARKER CREEK
13
14
15
16
EASTERN
FREEWAY
CALLINGTON
RIVER
BREMER
18
17
19
20
0km
2
4



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Nav eri
Coc pla
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Nit aur
Nav per
Nav ato
Mel var
Pin div
Stl pin
Tab fas
Nit inc
Nit agn
Sta ven
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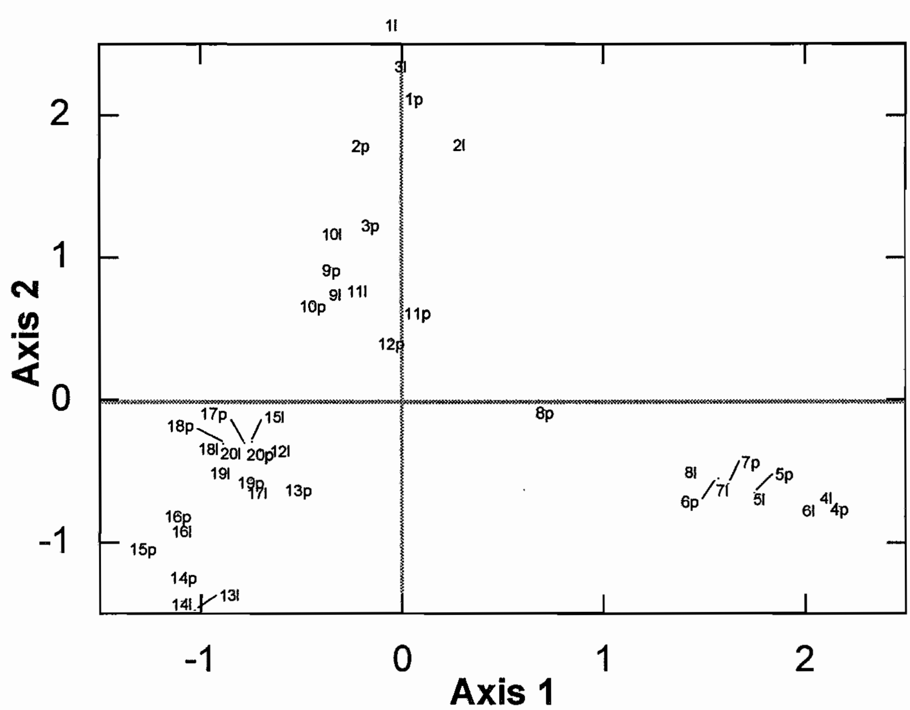

Axis 2
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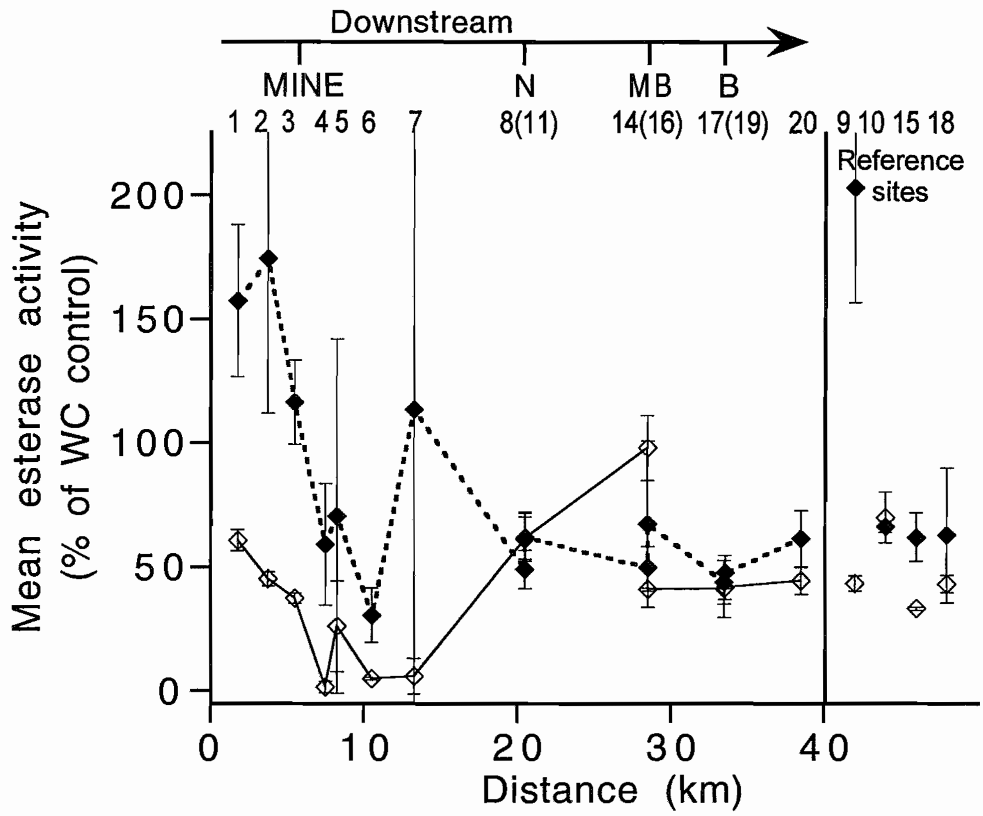
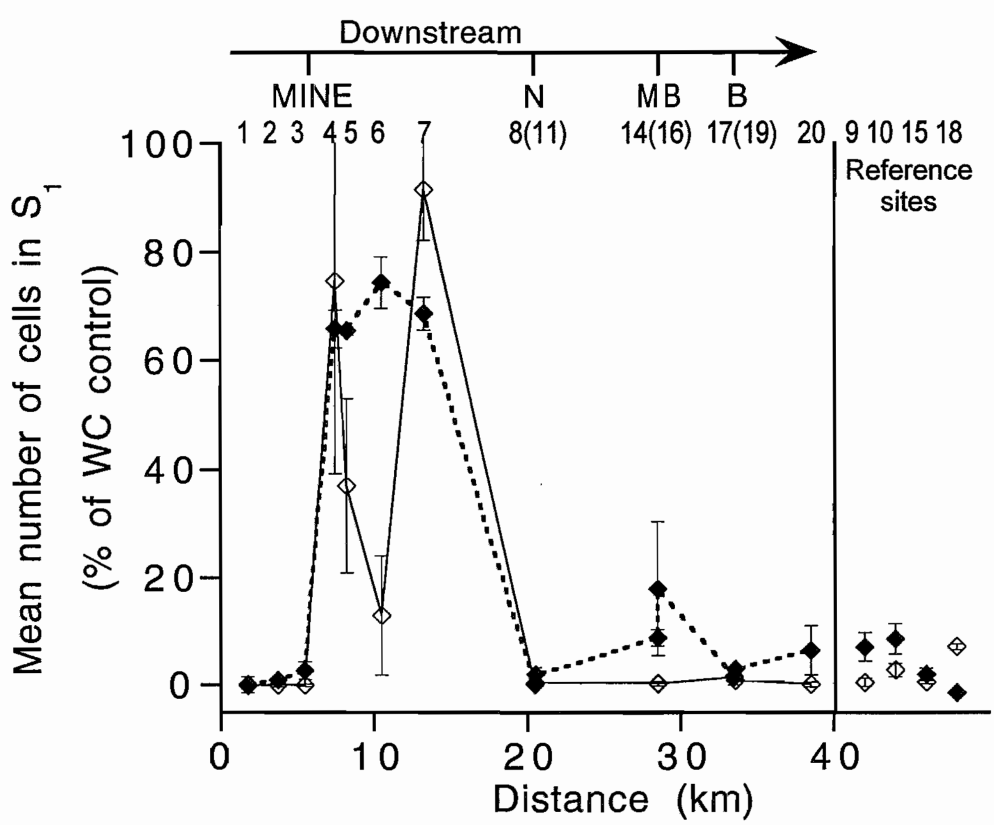
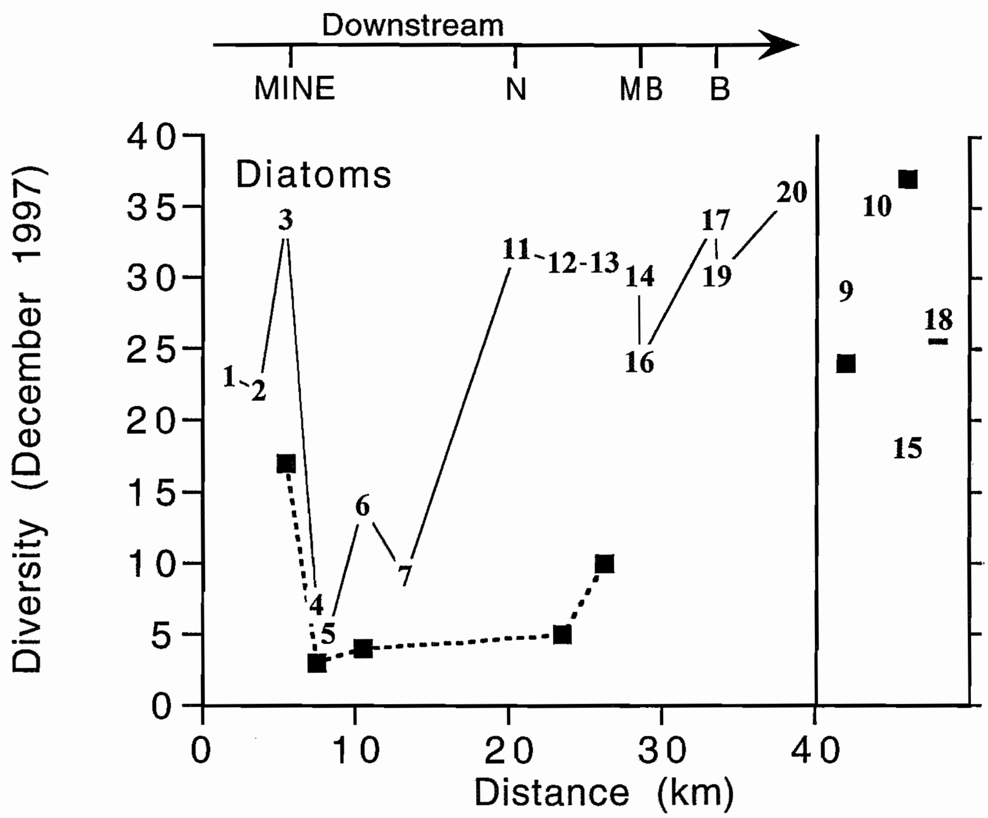

Downstream
MINE
N
MB
B
1 2 3 4 5 6 7
8(11)
14(16) 17(19) 20
9 10 15 18
Reference sites
Mean esterase activity (% of WC control)
200
150
100
50
0
0
10
20
30
40
Distance (km)


Downstream
MINE
N
MB
B
1 2 3 4 5 6 7
8(11)
14(16) 17(19) 20
9 10 15 18
Reference sites
Mean number of cells in S1 (% of WC control)
100
80
60
40
20
0
0
10
20
30
40
Distance (km)


Downstream
MINE
N
MB
B
Diatoms
Diversity (December 1997)
Distance (km)
0
5
10
15
20
25
30
35
40
0
10
20
30
40
1
2
3
4
5
6
7
9
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11
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17
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20

Peter Schultz – Australian Water Quality Centre, SA Water

Brukunga acid mine drainage: The macroinvertebrate story

Introduction

The project, initiated in the 1996–1997 financial year, involves macroinvertebrate and chemical monitoring at sites on the Dawesley Creek–Bremer River system. It forms part of a South Australian EPA licence agreement for the management of the disused Brukunga pyrites mine. The principal aims are to detect the impact of acid mine drainage (AMD) on Dawesley Creek and to measure its downstream extent.

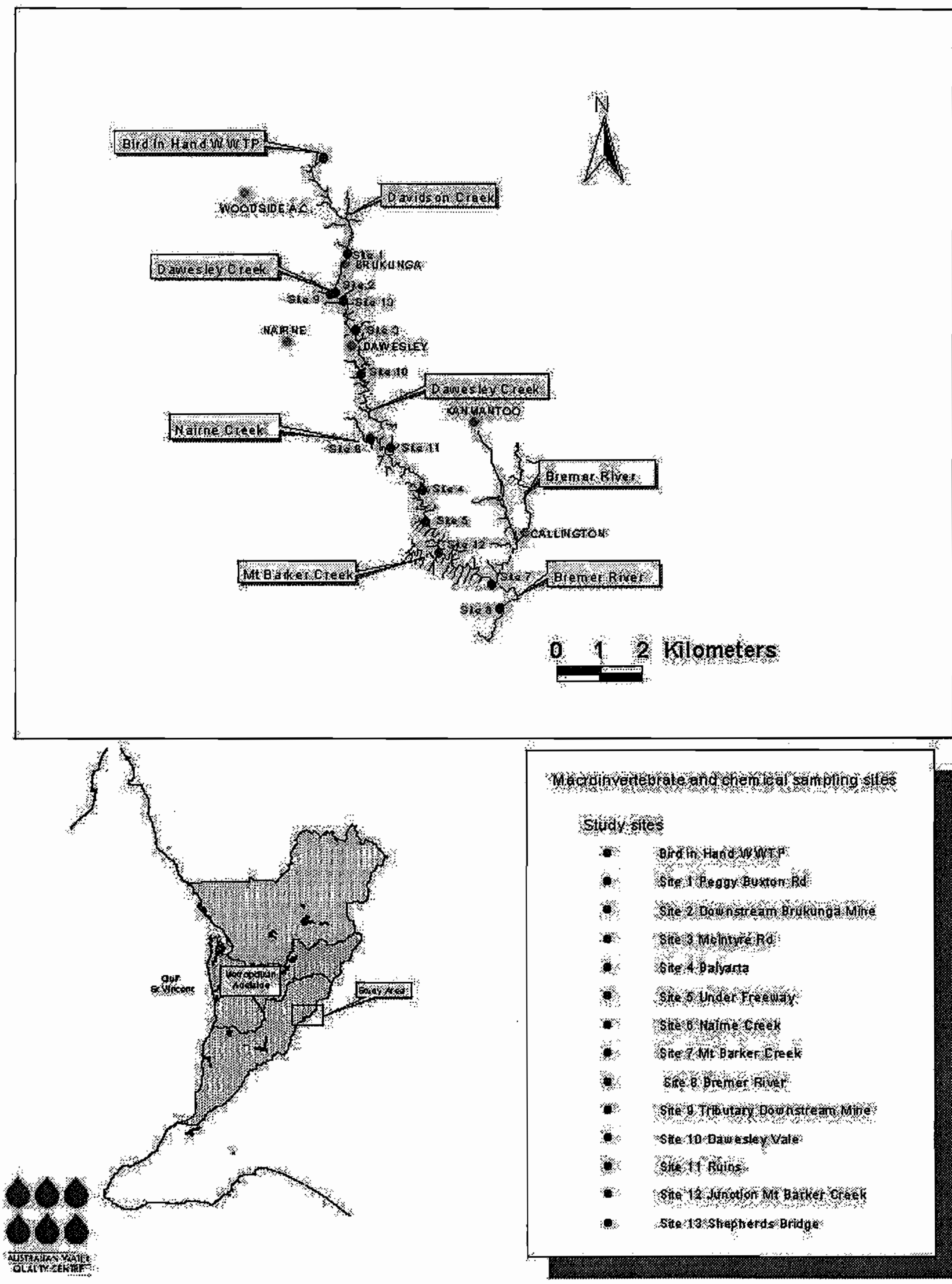
Site selection

The first sampling survey was carried out in September 1996 and coincided with an unusually large rainfall event that raised creek levels by as much as a metre above the normal winter/spring flows. The nine sites used were the same as a 1993 survey by Dr P. Suter at Australian Water Quality Centre. In December 1996, suitable weather and reasonable stream flow allowed the sampling team to access the stream at intermediate sites between Dawesley Creek and the confluence with Mount Barker Creek. Analysis of the December data lead to the adoption of six sites for continuous macroinvertebrate and chemical sampling and two sites further downstream for chemical sampling only. The sites are listed in Table 1 in the order they occur on the system. Project site numbers and the dates of macroinvertebrate sample collection are included. Figure 1 is a map of the streams showing all sites that have been sampled during the project.

Table 1. Project sites, site number and dates sampled.

Site name	Site	Sampling dates
Dawesley Ck at Peggy Buxton Rd	1	Sept 1996–Dec 1999
Dawesley Ck D/S Brukunga	2	Sept.1996–Dec 1999
Tributary just D/S Brukunga	9	Sept 1996
Dawesley Ck at Shepherds Bridge	10	Sept 1996 + Dec 1996
Dawesley Ck McIntyre Rd	3	Sept 1996–Dec 1999
Dawesley Ck Dawesley Vale	11	Dec 1996
Dawesley Ck ruins at Djatadapeel	12	Dec 1996
Nairne Ck at Djatadapeel Ford	6	Sept 1996–Dec 1999
Dawesley Ck at Balyarta Ford	4	Sept 1996–Dec 1999
Dawesley Ck at Freeway	5	Dec 1996–Dec 1999
Dawesley Ck U/S Mt Barker Ck	13	Sept 1997
Mt Barker Ck U/S Bremer River	7	Sept 1996–Dec 1999
Bremer River U/S Jaensch Rd	8	Sept 1996–Dec 1999

FIGURE 1
SKETCH OF THE DAWESLEY CREEK SYSTEM



Numbers refer to the macroinvertebrate and chemical sampling sites

Macroinvertebrate sampling

All AWQC work has involved use of the standard Monitoring River Health net, of 250 µm mesh, and sampling protocol where a 10 m section of stream is sampled using the standard method for edge habitat (CEPA 1994). Edge habitat was chosen because it is the only habitat always present, allowing for a fair comparison across all sites at all times. The study used a laboratory scoring protocol counting all organisms in a randomly chosen 10% of the sample from a Marchant sub-sampler (Marchant 1989). Identification is taken to the lowest practical level using the latest keys (Hawking 1999) and the AWQC voucher collection, the State's most extensive for freshwater macroinvertebrates. Identification was carried out at family level or lower in all groups except Turbellaria, Hydracarina, Nematoda and Oligochaeta. The identification for most Insecta and Mollusca is taken to species. The Dipteran families Ceratopogonidae, Tipulidae, Psychodidae, Stratiomyidae and Simuliidae are taken to species and the Culicidae are taken to genera. The Chironomidae are taken to tribe, genus or species depending on available keys while the other Dipteran families are not identified below Family.

Interpretation of macroinvertebrate data

Species richness, or the total number of different types of macroinvertebrate species identified in a sample, is still the most widely used statistic from community data to estimate the health of a stream in South Australia. The interpretation of derived statistics like diversity indices has not been able to provide a more consistent and meaningful measuring stick for stream health (Norris and Georges 1993).

Community composition

The species present in each sample and the abundance of each taxa reflects the differences between sites, habitats and conditions prior to the date of sampling. There are no simple rules for interpreting these complex data sets. Variation due to factors such as extreme flow events, time of day, chance colonisation by rare taxa, and weather conditions is always present to add to the differences due to water quality, season, habitat variability, ambient temperature and plant productivity. Testing of the variation in South Australian aquatic communities has been insufficient to create a definitive list of taxa that indicate gradation in water quality although AWQC experience can easily define the extremes. A pristine site has a vastly different community to polluted sites. Acid mine drainage reduces the community drastically. Only one species of chironomid, *Tanytarsus fuscithorax* is abundant in the low pH water. Very small numbers of predatory adult beetles and true bugs are also encountered. These air breathers are able to fly large distances and probably colonise the area during the preceding night. Eutrophic sites enriched by sewage discharge or agricultural practices have large numbers of nematode and oligochaete worms, and certain Chironomid species that are red because they contain haemoglobin. Large numbers of immature bugs of several species are also present. If oxygen concentrations remain high throughout the 24 h of each day, many other species are also found and species richness can be inflated.

Macroinvertebrate results

The actual results consist of a list of the taxa recorded at each site, the species richness, and the abundance estimate of each taxon present. In this report, only the most useful statistic, species richness, is presented and discussed in detail. The species richness measures the diversity of the macroinvertebrate community. A large number of species finding the conditions suitable indicates a more natural site with less human induced stress. It is also accepted that a more diverse community will recover more quickly from both natural and human induced stresses compared with a community with low species richness.

Longitudinal changes in the stream

Mean results

Figure 2 shows the mean species richness and the standard deviation for each site arranged in order on the stream. The first site at Peggy Buxton Road is upstream of the Brukunga mine. It is not a good measure of an unimpacted site because it is eutrophic due to the Bird in Hand wastewater treatment plant at the headwaters of Dawesley Creek. Nairne Creek and Tributary 1 are other control sites. The mean results show the clear impact of AMD on Dawesley Creek. From the mine to the confluence with Nairne Creek, just downstream of the ruins at Djatadapeel, species richness is always reduced. Between the confluence with Nairne Creek and Mount Barker Creek the effect is variable with time or flow. This is shown by the higher mean and the high standard deviation at the two sites, Dawesley Creek at Balyarta and at the Freeway. The impact downstream of Mount Barker Creek is much less with species richness approaching that of the control site on Nairne Creek.

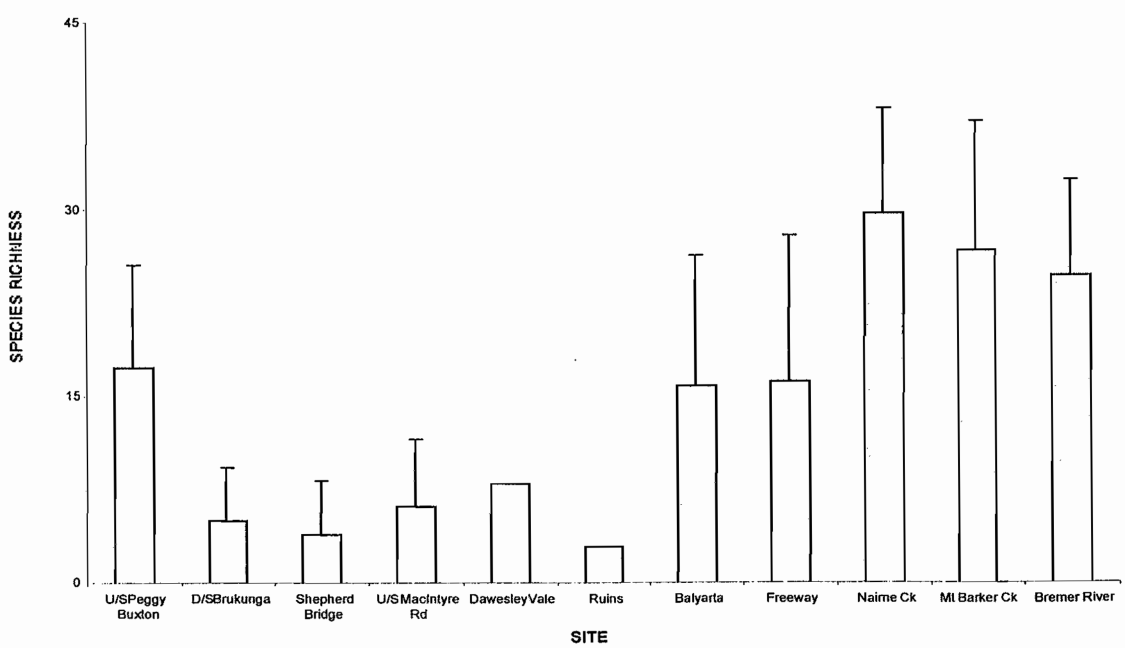


Figure 2. Mean and standard deviation for species richness at each site (September 1996 to December 1998).

September 1996

This event was unique in the study. The sites were chosen to duplicate the sampling event of 1993 with the intention of adding additional sites if a rapid scan of each sample showed a rapid change in water quality between two sites. The event was also programmed as a training exercise for personnel from another agency, which locked the date into the calendar. The sampling went ahead despite overnight rain and the threat of more heavy falls during the day. The results are atypical of the stream but provided useful data about extreme flow events and assisted interpretation of other sampling events. Figure 3 graphs the species richness and pH against site. The pH downstream of the mine was the highest recorded during the study. The species richness at the sites down stream of the mine is higher than that measured at any other time. This resulted from the rapid flow translocating animals from sites upstream of Peggy Buxton Road. At all other sites the species richness is reduced due to the high flow. The community composition still showed an impact of AMD.

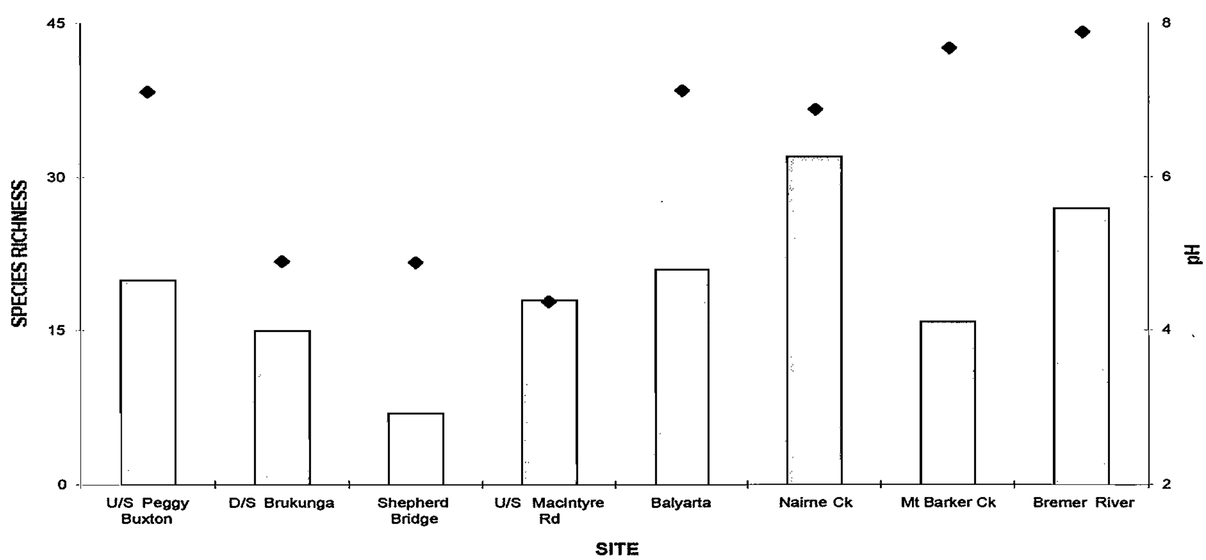


Figure 3. Species richness and pH (September 1996).

December 1996

Eleven sites were sampled including three new sites on Dawesley Creek. These were at Dawesley Vale, the ruins at Djatadapeel, just upstream of the confluence with Nairne Creek, and under the Freeway (Figure 4). The pH suggested that AMD extended with minimal dilution to the confluence with Nairne Creek but not beyond. Species richness in the section of Dawesley Creek downstream of the mine as far as Nairne Creek was severely reduced. By the Freeway site the community composition had similar species richness to the control site. The reduced species richness at Mount Barker Creek and the Bremer River site was attributed to the very different river morphology at these sites due to the higher stream order and flow. This survey determined the sites for all subsequent surveys. It seemed unnecessary to continue sampling macroinvertebrates downstream of the confluence with Mount Barker Creek. Access to sites on Dawesley Creek between McIntyre Road and Nairne Creek is difficult and these were also omitted.

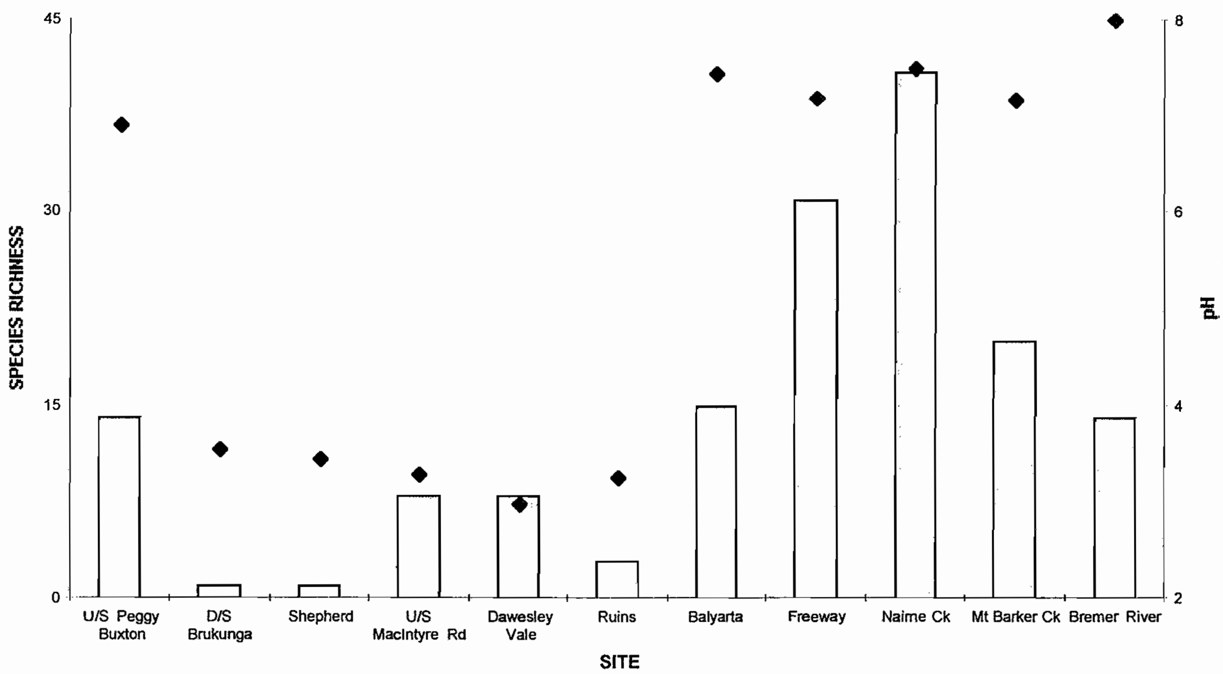


Figure 4. Species richness and pH (December 1996).

March 1997

This autumn sampling had a flow throughout Dawesley Creek to the ford at Balyarta (Figure 5). The impact of the mine is clear upstream of Nairne Creek. By Balyarta the species richness is improving. The Freeway site was dry.

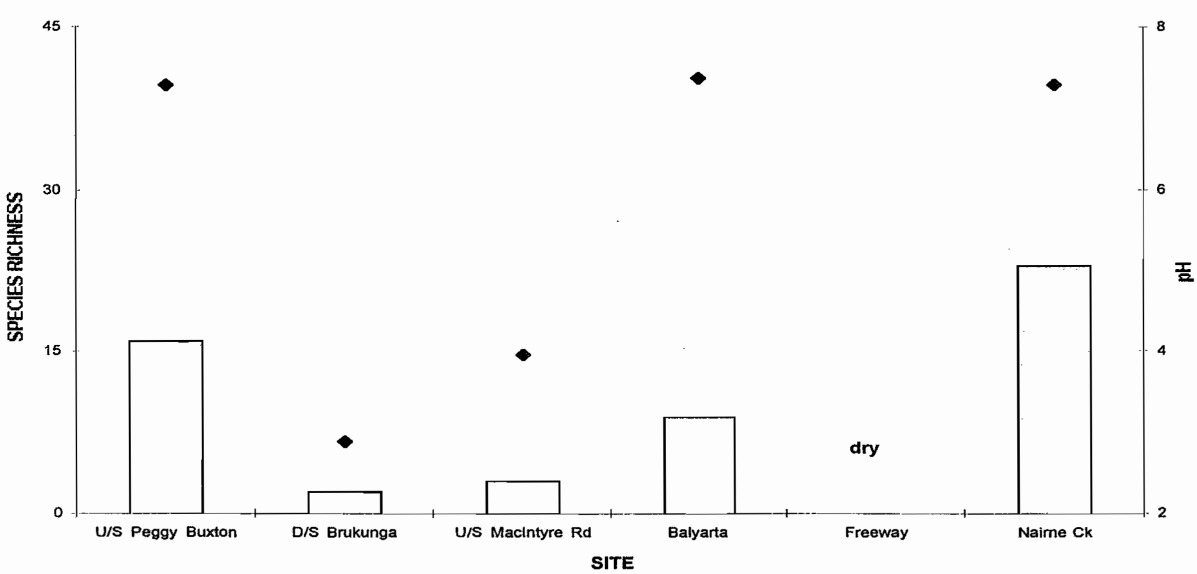


Figure 5. Species richness and pH (March 1997).

June 1997

Species richness was reduced at Peggy Buxton Road and was associated with low dissolved oxygen. Figure 6 clearly shows that the results from December 1996 were repeated. Acid mine drainage impact occurs at both sites upstream of Nairne Creek with rapid recovery downstream of the confluence with Nairne Creek. Mount Barker Creek and the Bremer River were included in the Monitoring River Health sampling strategy, so data for these two sites are included. Species richness clearly follows pH: pH increased downstream of the confluence with Nairne Creek, suggesting that significant dilution of water occurs at the confluence of Dawesley Creek.

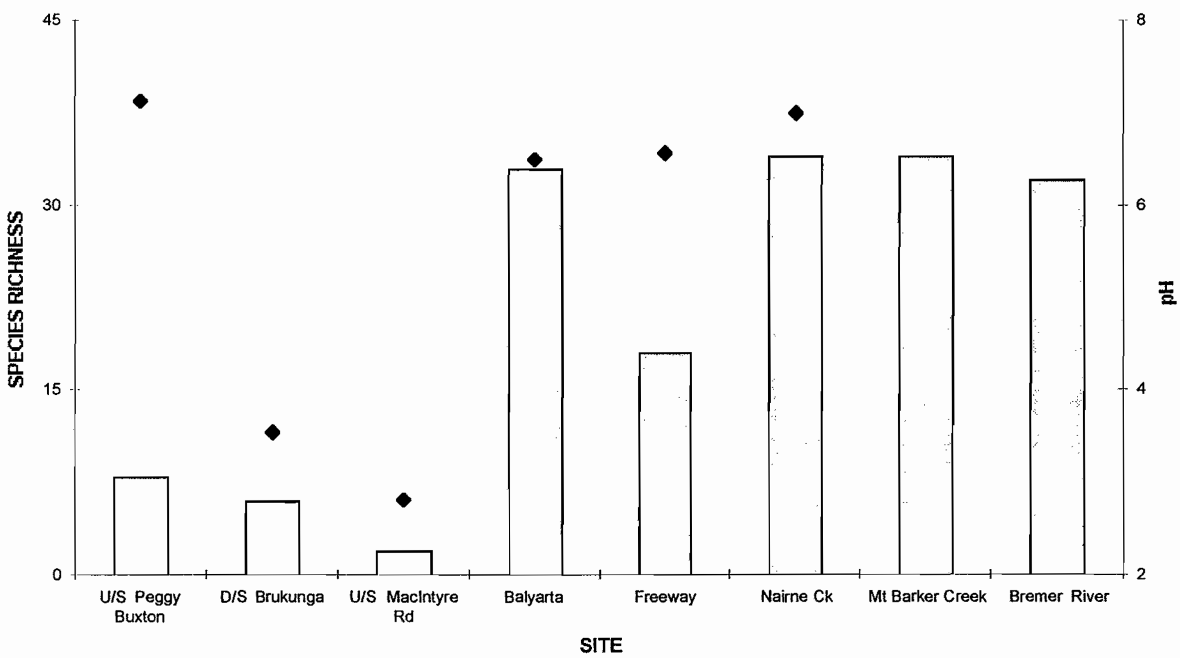


Figure 6. Species richness and pH (June 1997).

September 1997

Increased rainfall in early Spring produced a significant change in the stream condition in September (Figure 7). It was clearly visible during the sampling that Dawesley Creek downstream of the confluence with Nairne Creek was different and that the difference was a recent event. Extensive filamentous algae mats had developed, but they were an unusual green—almost looking cooked. Other macrophytes, especially *Triglochin procerum,* also appeared stressed. The pH was 4.31 at the Mine site but it remained acidic right to the Freeway site (pH = 4.69) where the previous water samples had been alkaline. Species richness was reduced at Balyarta and the Freeway site compared with previous samples.

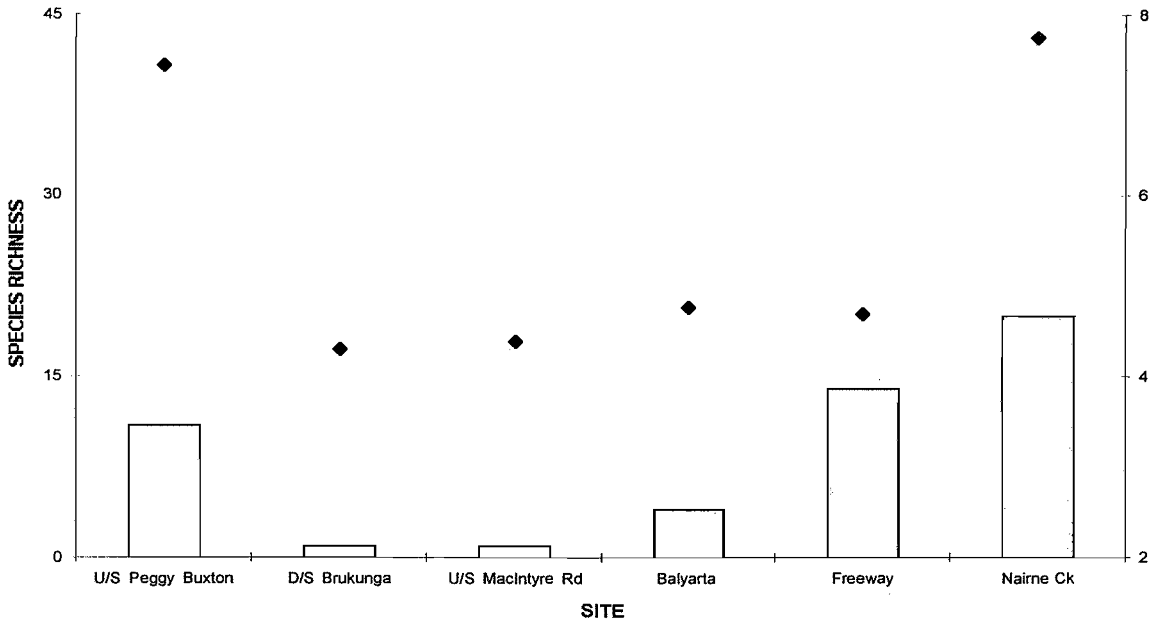


Figure 7. Species richness and pH (September 1997).

December 1997

High flows were maintained throughout Spring and the impact of AMD was detectable in both species richness and pH for the whole length of Dawesley Creek (Figure 8). Monitoring River Health data was available to show that Mount Barker Creek and the Bremer River sites had normal species richness when compared with the control site on Nairne Creek.

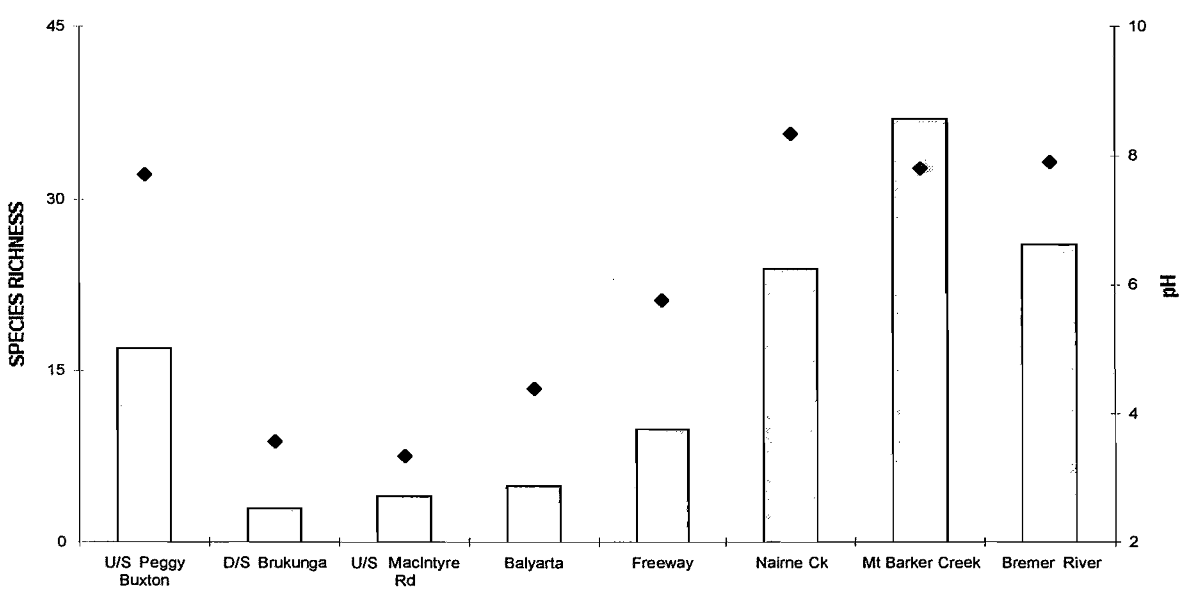


Figure 8. Species richness and pH (December 1997).

March 1998, June 1998, September 1998 and December 1998

The pattern of 1997 was repeated in 1998 (Figures 9–12). Dawesley Creek was dry above Nairne Creek in March. By June the AMD impact reached only to Nairne Creek and by September the impact was detectable to Mount Barker Creek (Figure 11).

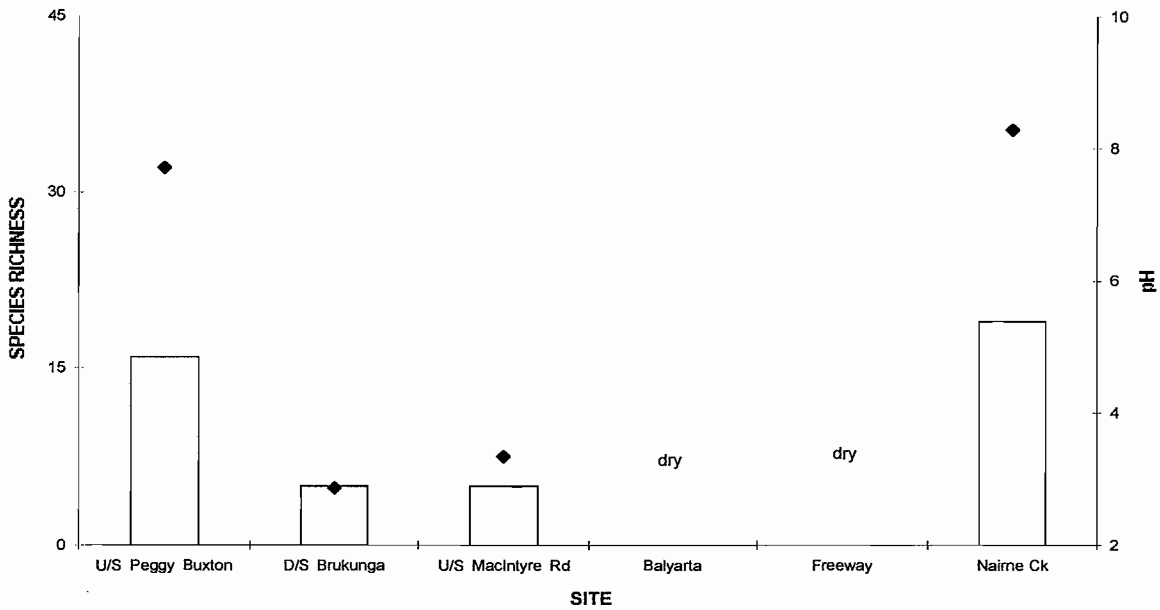


Figure 9. Species richness and pH (March 1998).

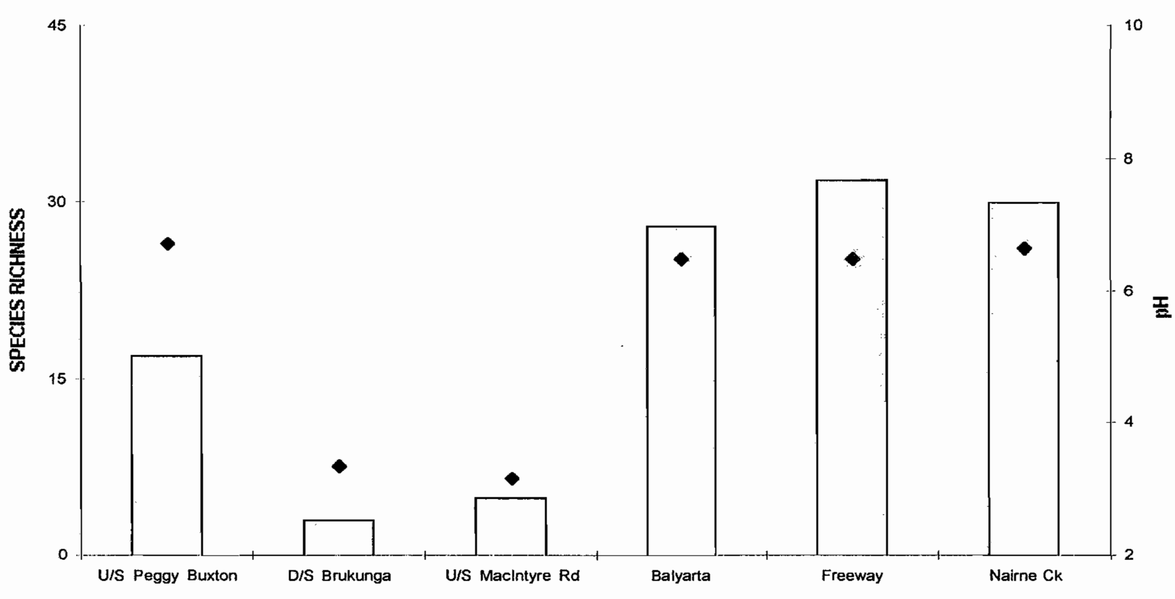


Figure 10. Species richness and pH (June 1998).

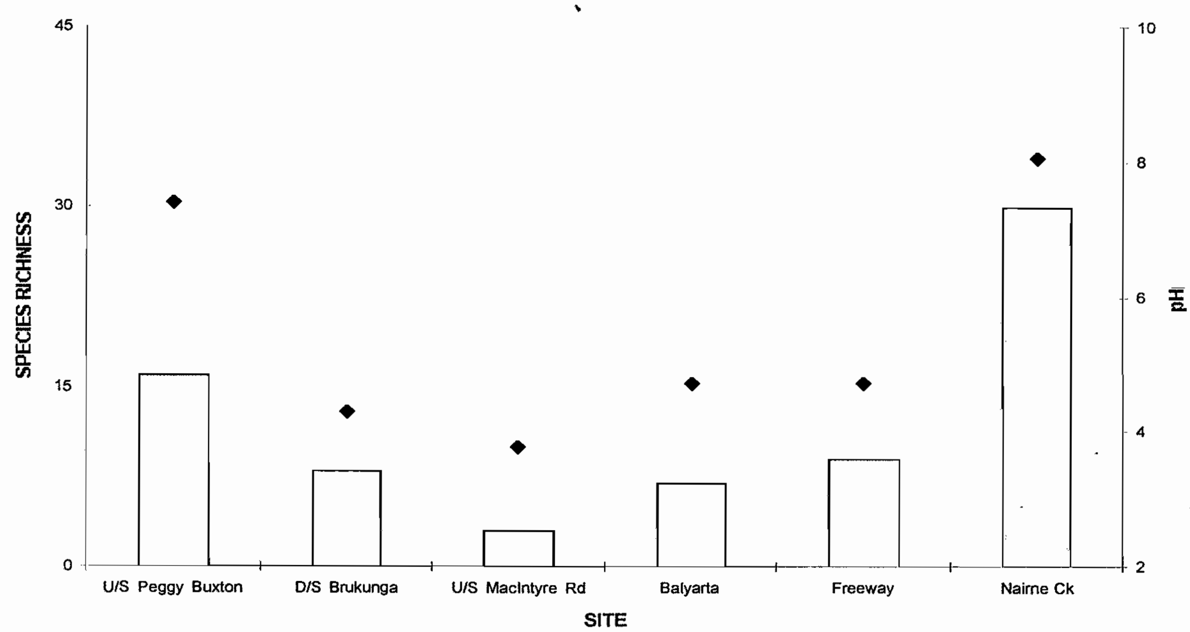


Figure 11. Species richness and pH (September 1998).

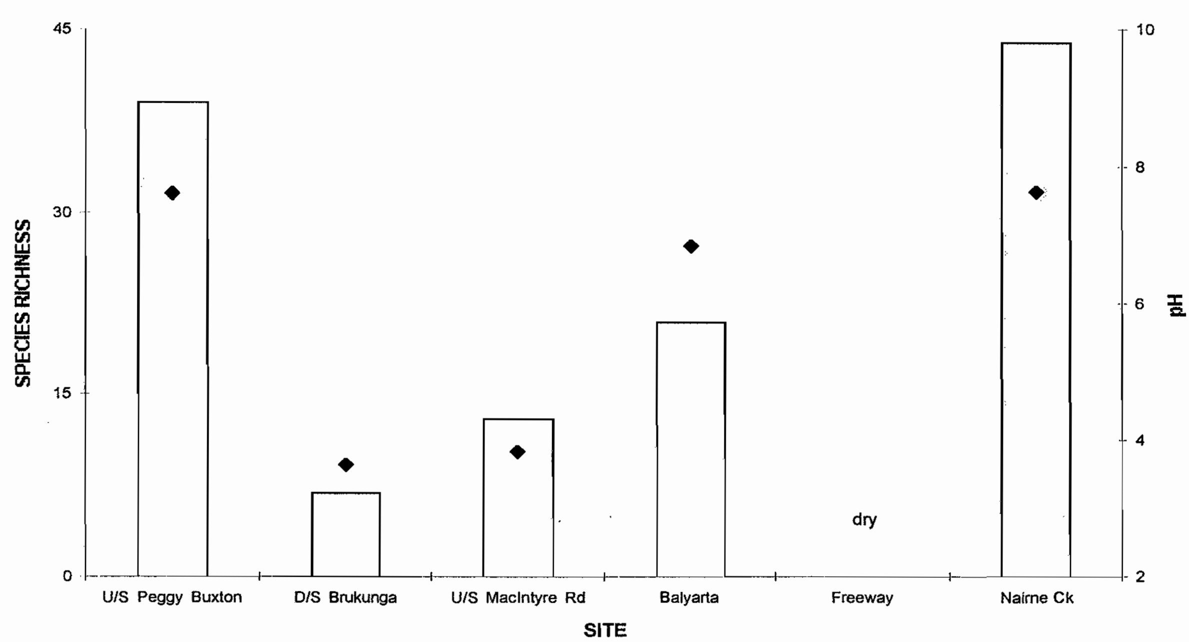


Figure 12. Species richness and pH (December 1998).

Temporal changes at each site

Most sites presented a consistent pattern throughout the study. The two sites on Dawesley Creek downstream of Nairne Creek, at Balyarta and the Freeway, are the exceptions. These two sites are often dry in late summer and autumn. Late autumn and early winter flows are neutral to slightly alkaline but the increased flow in late winter and spring is acid.

Peggy Buxton Road

The Wastewater Treatment Plant influences conditions at Peggy Buxton Road. Species richness is depressed at times due to eutrophic conditions and low dissolved oxygen (DO) concentrations DO (Figure 13). pH is normal for a Mount Lofty Range stream. Flow is variable and can cease in autumn.

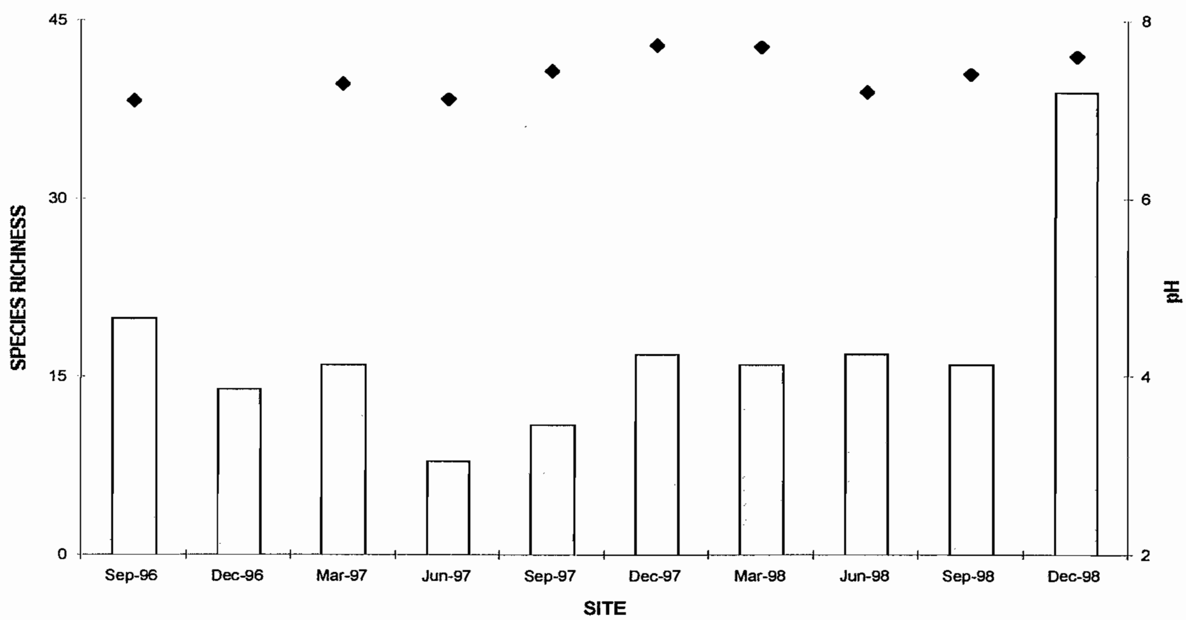


Figure 13. Species richness and pH at Peggy Buxton Road.

Downstream Brukunga

Acid mine drainage with low pH discharges throughout the mine site. Higher pH values were obtained each September with elevated flows. Species richness remains depressed at all times (Figure 14).

McIntyre Road

A similar story for pH at this site (Figure 15) during higher flows. An increase in pH in March 1997 compared with downstream Brukunga occurred at low flow.

Balyarta

The sampling in September and December 1996 gave no indication of the pattern in the next two years (Figure 16). Alkaline pH values in autumn and early winter changed to acidic conditions by early spring with an associated drop in species richness. The period that pH remains low varies from year to year and is probably dependent on rainfall and hydrology.

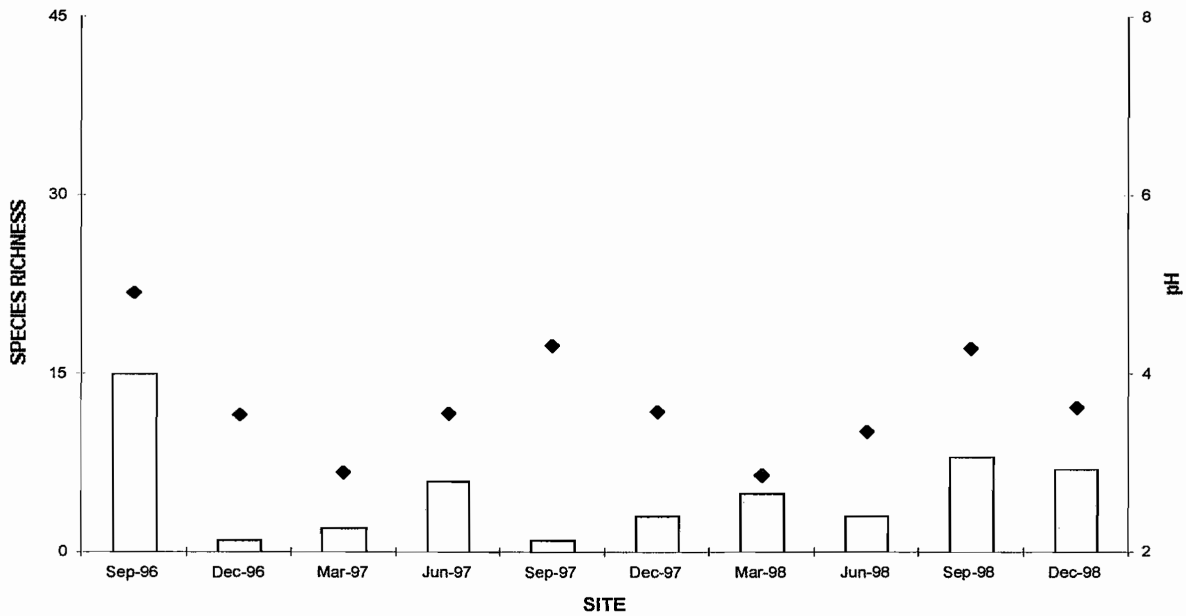


Figure 14. Species richness and pH downstream of Brukunga mine.

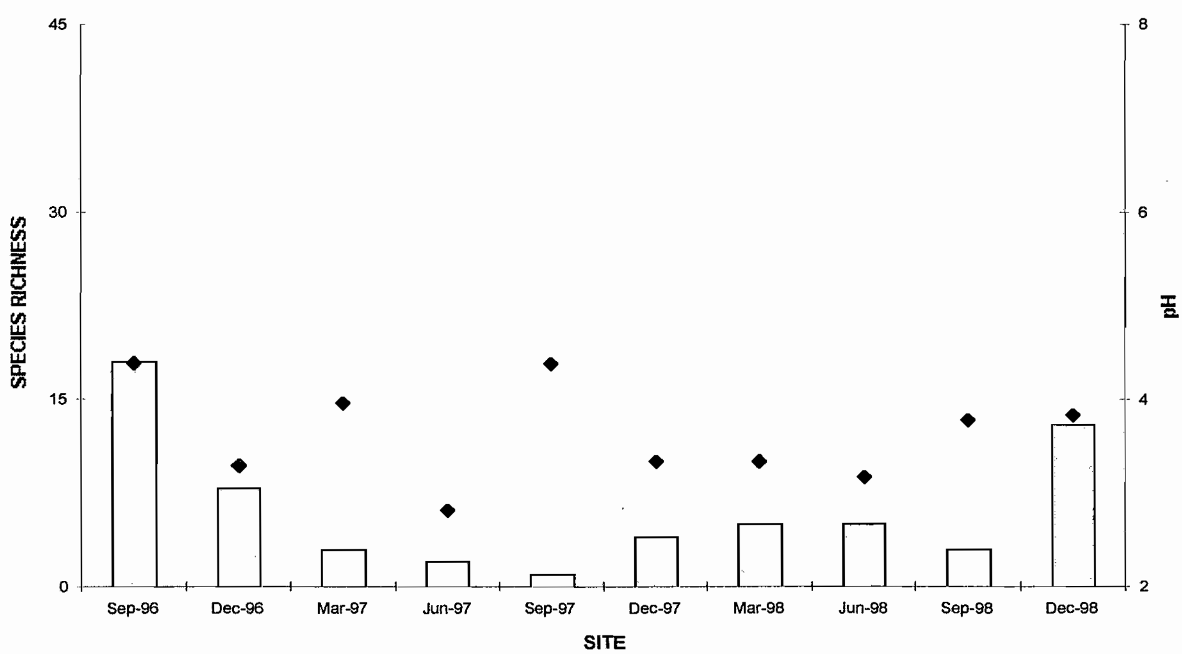


Figure 15. Species richness and pH at McIntyre Road.

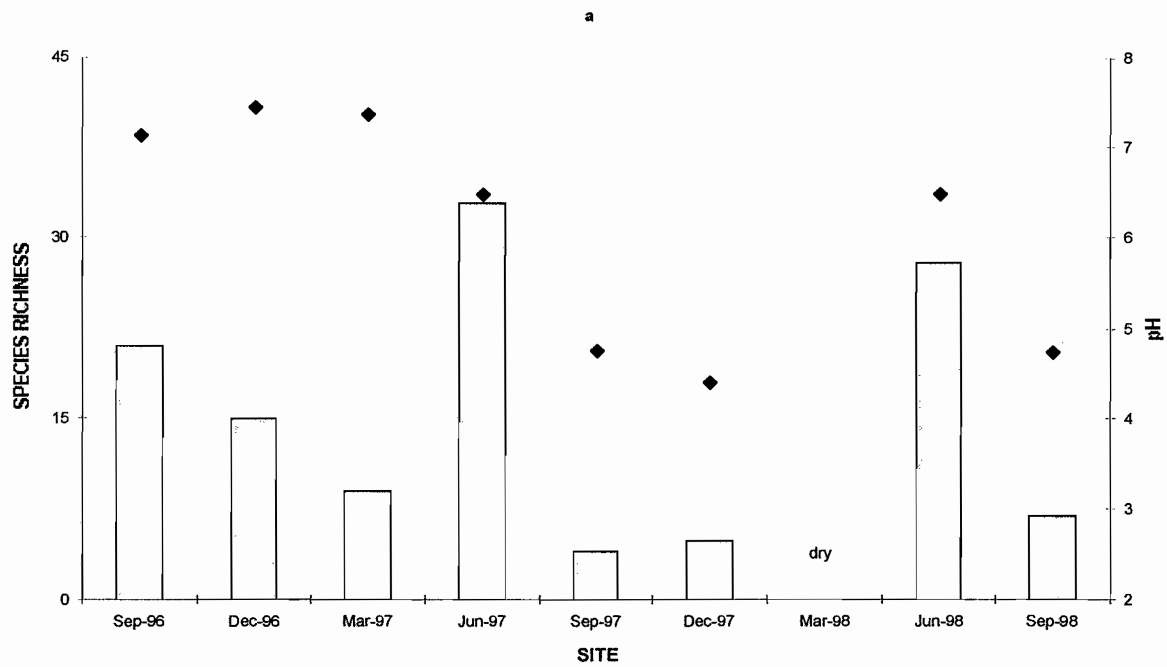


Figure 16. Species richness and pH at Balyarta.

Freeway

Reduced pH at high flow in the spring of 1997 and 1998 is also apparent here (Figure 17). The site also dries up earlier than Balyarta each summer. Flows from summer storms do not remain for long. Species richness is always low in early winter, as recolonisation must occur.

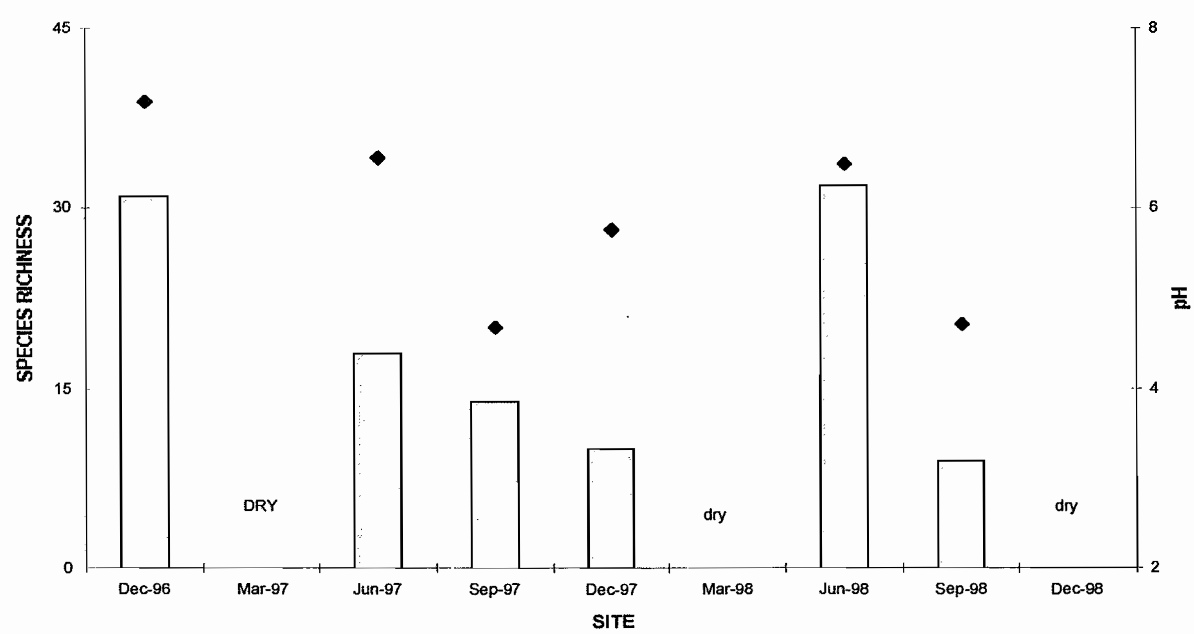


Figure 17. Species richness and pH at Freeway.

Nairne Creek

The control site is slightly eutrophic but has a species richness and pH consistent with Mt Lofty Range streams. It reflects the condition we would expect in Dawesley Creek if AMD and the influence of Bird in Hand wastewater treatment plant were not present. There is a drop in species richness each autumn and after a flood event in August 1997 (Figure 18).

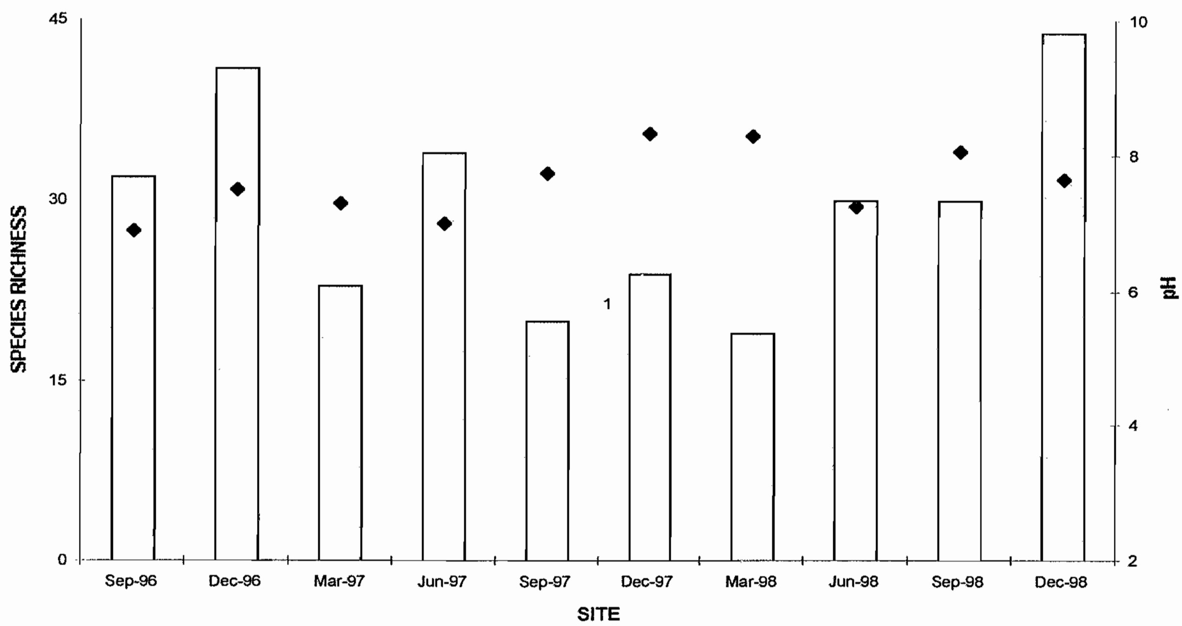


Figure 18. Species richness and pH at Nairne Creek.

Changes in the Al/Fe ratio

The mine produces a huge increase in the concentration of iron and aluminium in Dawesley Creek. The pre-mine concentrations in Dawesley Creek (Figures 19 and 20) and in Nairne Creek (Figure 21) are two orders of magnitude lower than at the site immediately downstream of the mine (Figure 22). The concentrations of both metals decreases downstream as precipitation of the metals occurs–the iron concentration decreases more rapidly than the aluminium concentration (Figures 23–25). The ratio of aluminium to iron changes, increasing downstream. At the two sites downstream of Nairne Creek a seasonal peak in the Al:Fe ratio occurs in late winter, through spring to early summer (Figures 24 and 25). This coincided with the measured seasonal reduction in pH and species richness at these sites.

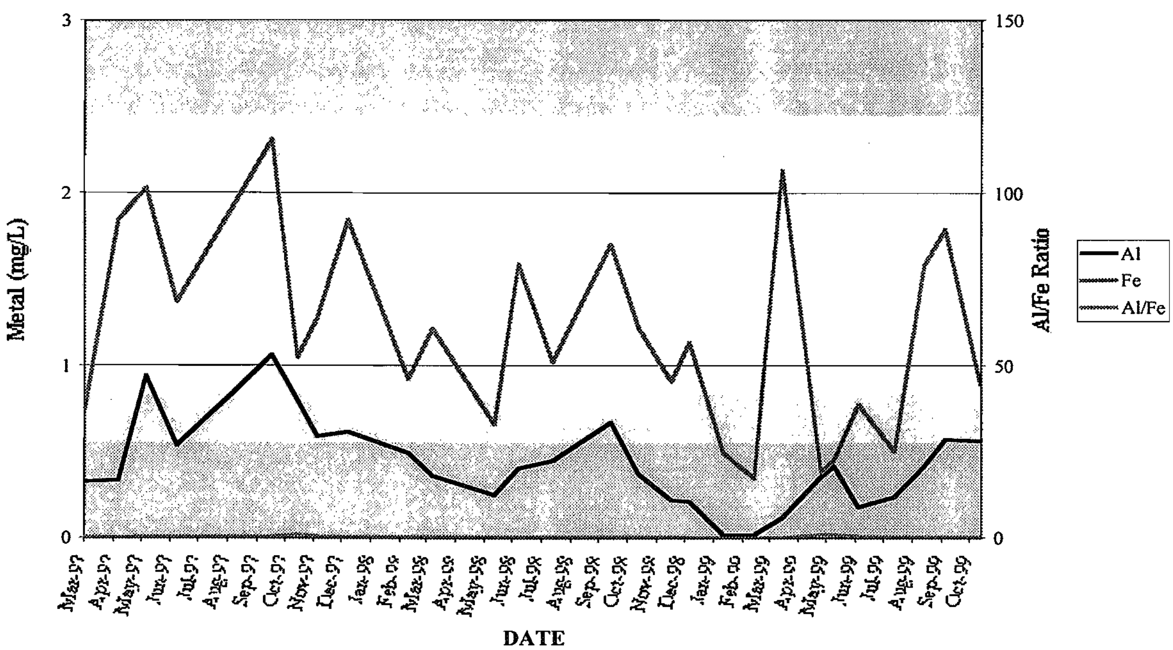


Figure 19. Concentration of Al and Fe and the Al/Fe ratio at Peggy Buxton Road.

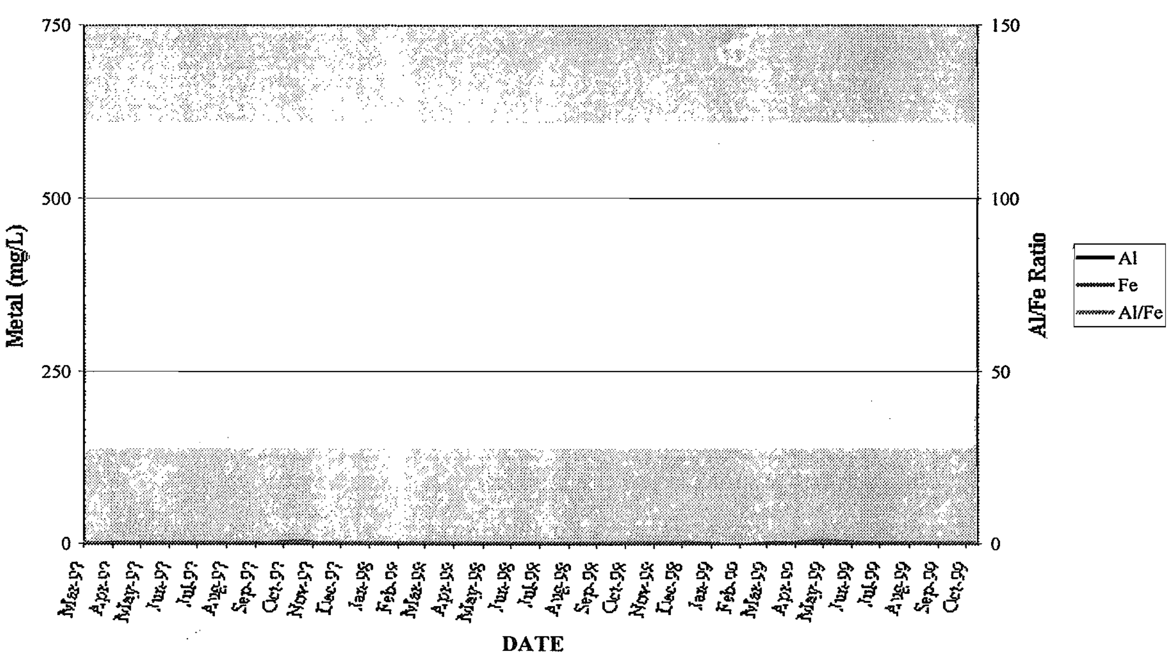


Figure 20. Concentration of Al and Fe and the Al/Fe ratio at Peggy Buxton Road scaled to highlight the differences in downstream concentrations/ratios.

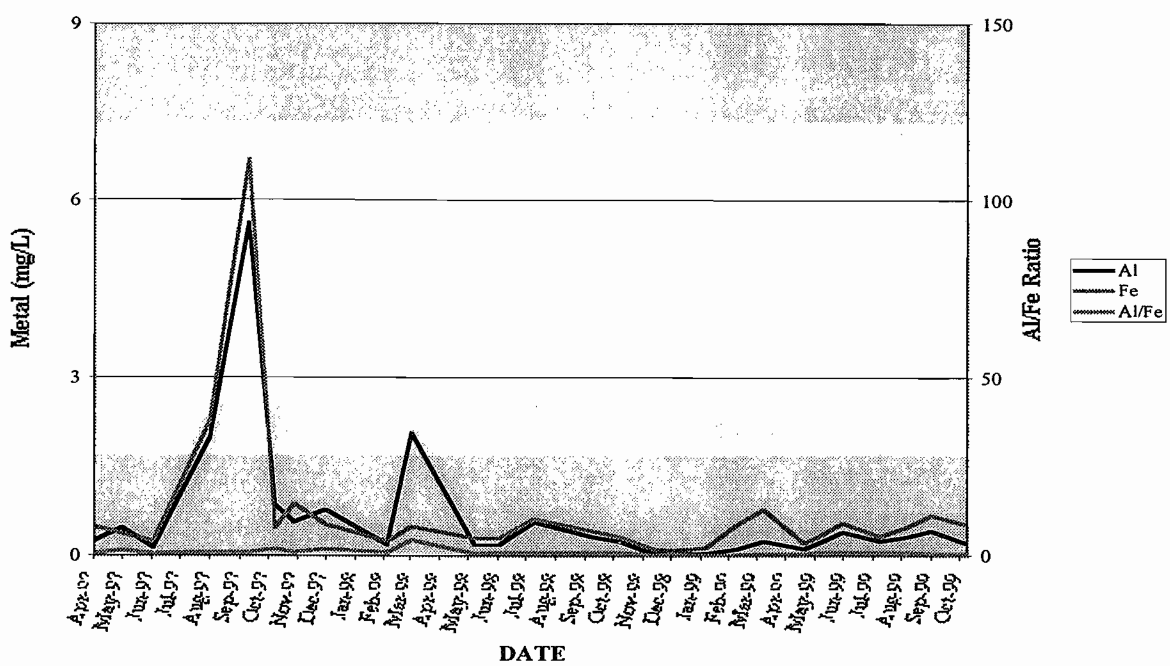


Figure 21. Concentration of Al and Fe and the Al/Fe ratio at Nairne Creek.

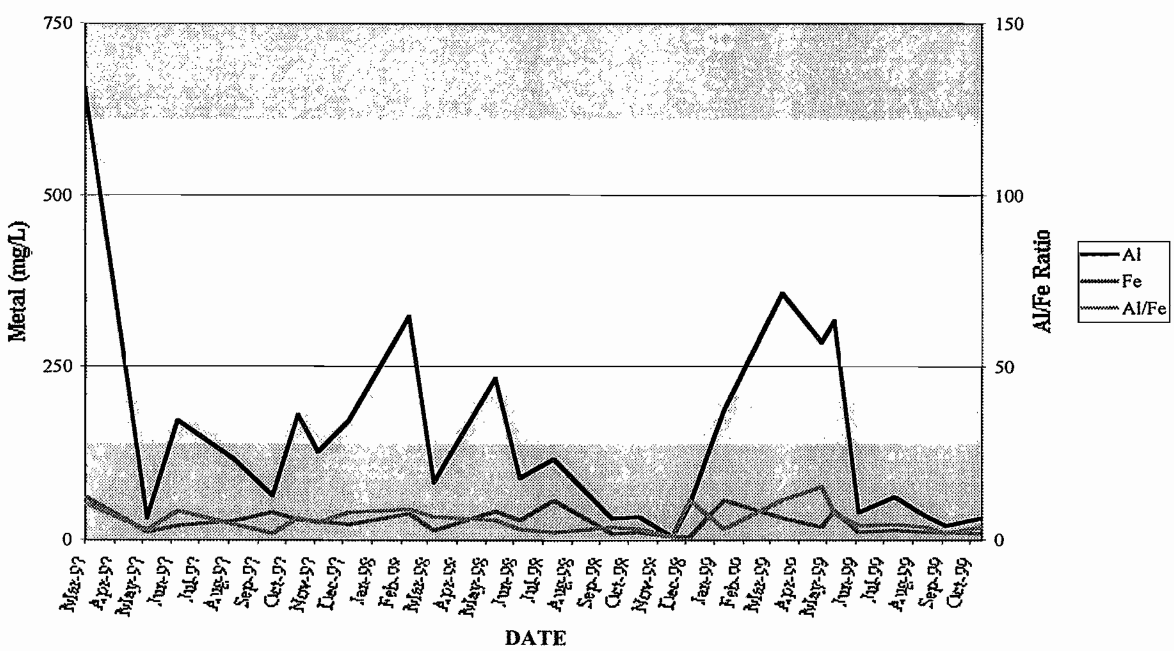


Figure 22. Concentration of Al and Fe and the Al/Fe ratio downstream of Brukunga.

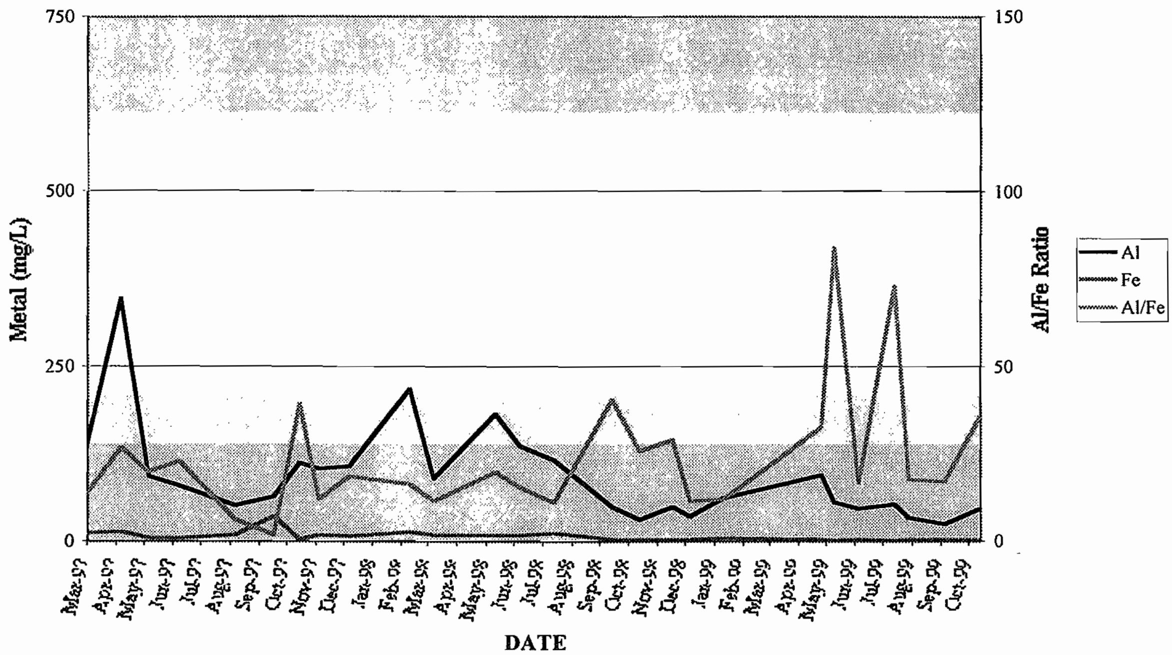


Figure 23. Concentration of Al and Fe and the Al/Fe ratio at McIntyre Road.

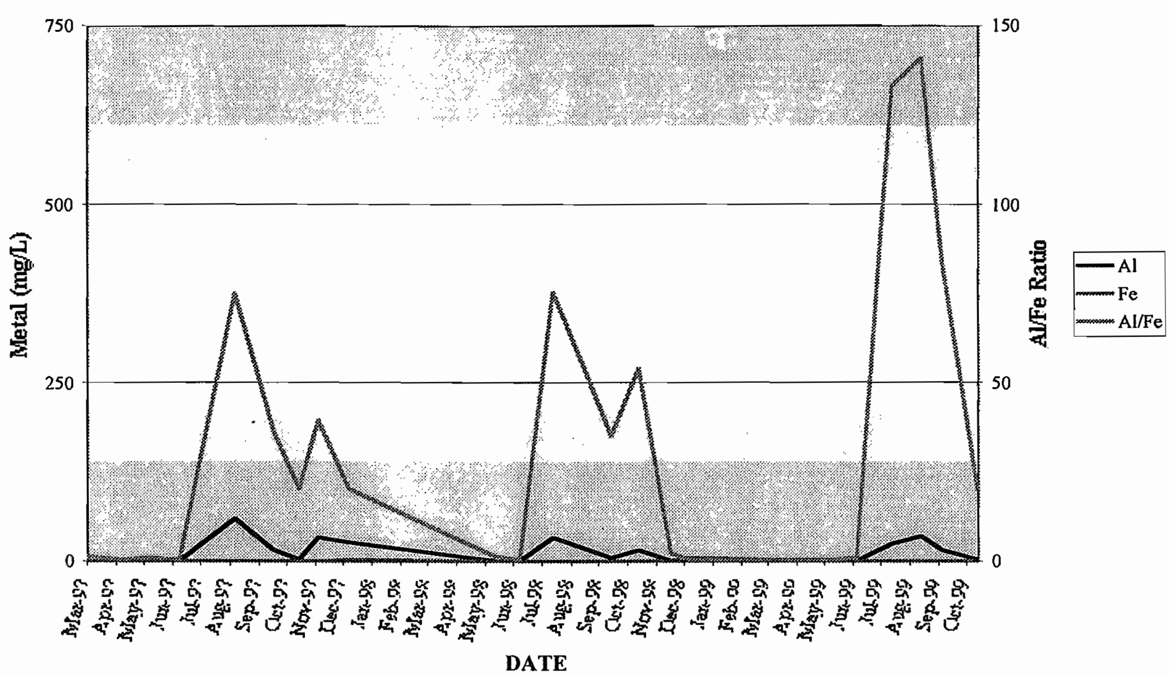


Figure 24. Concentration of Al and Fe and the Al/Fe ratio at Balyarta.

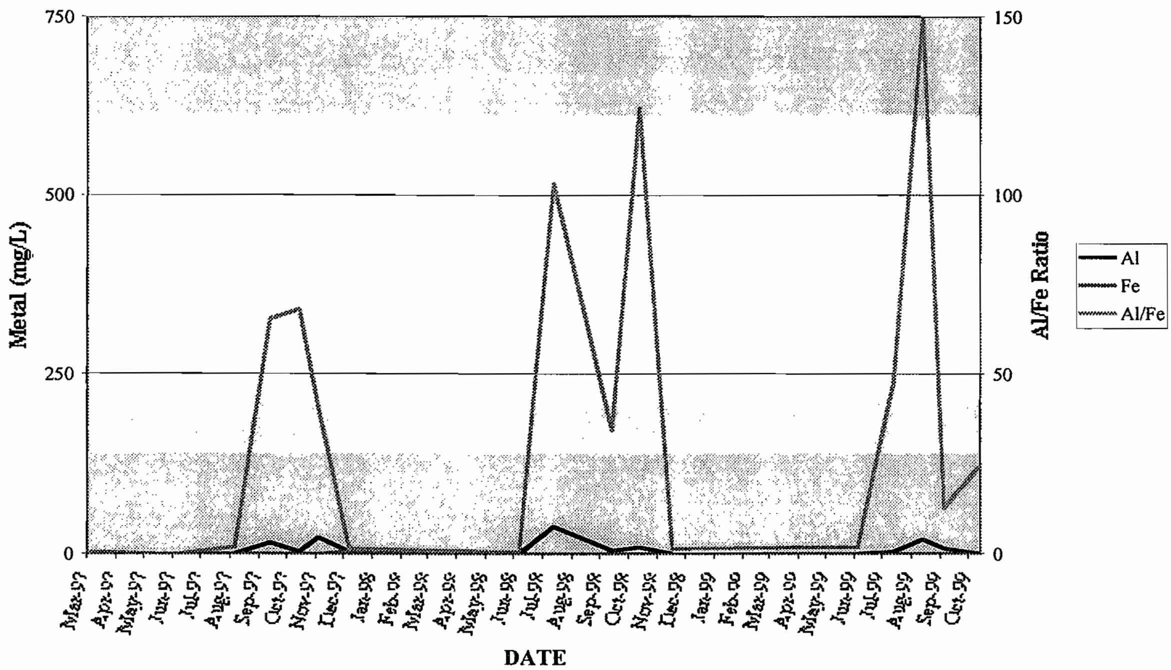


Figure 25. Concentration of Al and Fe and the Al/Fe ratio at Freeway.

Discussion

Despite treatment of as much of the groundwater from the mine site as possible with the present collecting system, the water in Dawesley Creek is highly acidic and unsuitable as habitat for most macroinvertebrates. The downstream extent of the impact reaches to the confluence of Dawesley and Nairne Creeks at all times except when extreme flood discharges dilute the system for brief periods. The seasonal fluctuation in flow in Dawesley Creek produces a seasonal fluctuation in the pH and macroinvertebrate community downstream of the confluence with Nairne Creek. At low flow, Nairne Creek dilutes the discharge from Dawesley Creek so the sites at Balyarta and the Freeway have a 'normal' community and pH. This occurs during the early winter and through summer. Late summer and autumn flows are minimal downstream of Nairne Creek. However, during the months of highest base flow, low pH water extends as far downstream as Mount Barker Creek. The impact on the relatively intact community in this lower section of Dawesley Creek is quite severe.

The cause of the variation must be linked to hydrology of groundwater near the mine site. The proportion of the flow from Dawesley Creek upstream of the confluence with Nairne Creek must increase and simple rainfall increases should not produce a change in proportional discharge. It is possible that rain in early winter increases the head on the acid ground water in Brukunga and its flow increases steadily as the wet season progresses. This would change the proportion of AMD in the stream. Seasonal increases in the ratio of aluminium to iron downstream of the confluence with Nairne Creek are observed during the period of lower pH in this section of Dawesley Creek (Figures 24 and 25).

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John Twining – Managing Mine Wastes Project, ANSTO
Peter Schultz and Tim Thompson – Australian Water Quality Centre, SA Water

Application of AQUARISK to the Dawesley Creek system using water quality and macroinvertebrate monitoring data: Implications for ecosystem rehabilitation strategy

A computer program for ecological risk assessment (ERA), AQUARISK, has been developed and applied to evaluate the likelihood of biotic detriment at two sites in Dawesley Creek due to exposure to acid rock drainage (ARD) from the Brukunga mine site in the Adelaide Hills of South Australia (Fig. 1).

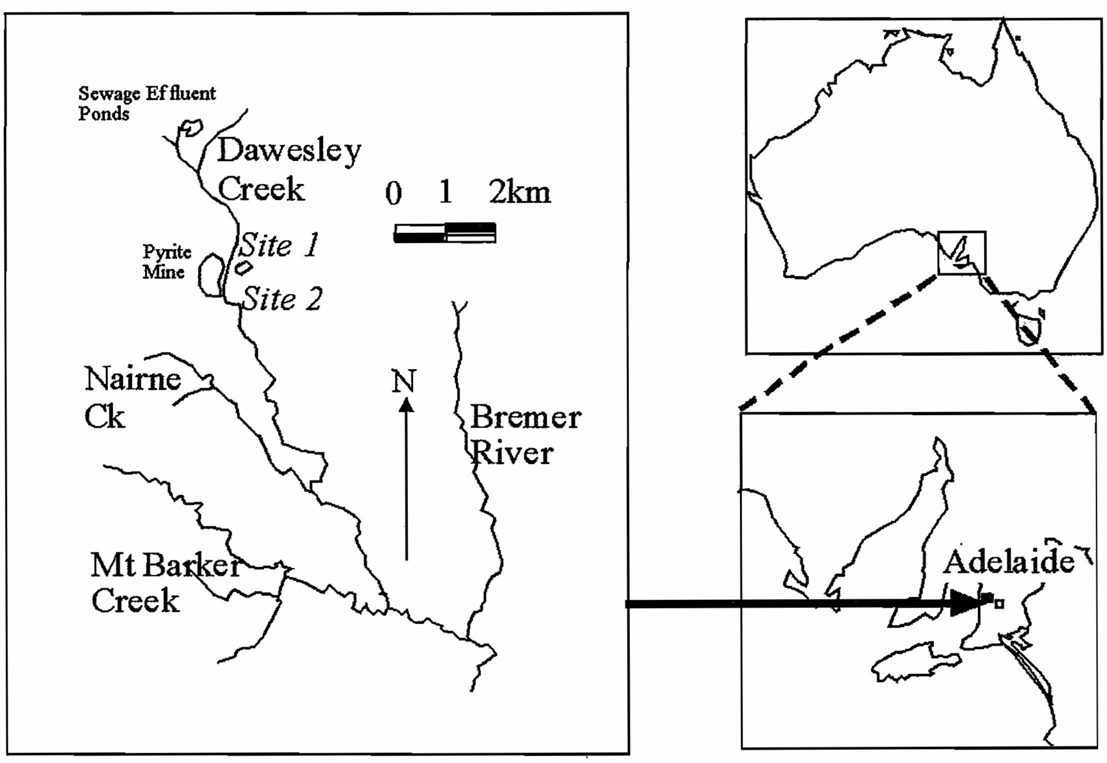

Figure 1. Site location map. Sites 1 and 2 are immediately adjacent to the Brukunga mine on Dawesley Creek.

Stream water quality monitoring data were used for the assessment and were compared with (a) national water quality guidelines, (b) literature derived ecotoxicology data and (c) biological monitoring data. The average monitored values for the major parameters in water at each of the sites are given in Table 1. It should be noted that individual measures, rather than averages, are used for the ERA.

Table 1. Average major cation and anion concentrations, total organic carbon (TOC), pH and conductivity values [mean (s.d.)] measured in samples collected at each site.

Site	pH	Conductivity µS/cm @25°C	Ca^{2+} mg/L	Mg^{2+} mg/L	Na^{+} mg/L	Cl^{-} mg/L	SO_4^{2-} mg/L	TOC mg/L
Upstream	7.7 (0.4)	2008 (561)	53 (23)	46 (20)	406 (178)	478 (210)	79 (35)	34
Downstream	3.8 (1.0)	3283 (1086)	150 (82)	103 (56)	307 (168)	620 (340)	1461 (801)	16

The ERAs were performed using a) the measured total concentrations of heavy metals at each of the sites and b) the bioavailable concentrations of some metals based on speciation modelling calculations performed within AQUARISK. The average values for the concentrations of the toxic metals at each of the sites is given in Table 2.

Table 2. Average concentrations of toxic metals at each of the sites as measured (total) and estimated (Bioavail.). Missing data indicate those metals for which the AQUARISK code does not currently evaluate chemical speciation.

Site	Metal Fraction	Cu µg/L	Mn µg/L	Ni µg/L	Pb µg/L	Zn µg/L	Cd µg/L	Al µg/L	Cr µg/L
Upstream	Total	8	150	6	1.3	83	<0.1	1100	6
	Bioavail	6	140		0.9	76	<0.1	3	
Downstream	Total	170	13000	530	4.7	8600	82	190000	39
	Bioavail	140	11000		3.5	6400	50	54000	

The results of macroinvertebrate monitoring for the two sites is shown in Figure 2 along with other sites in the system. Nairne Creek was chosen as the most appropriate reference but an average value was also derived using Nairne Creek plus three other sites that were apparently beyond the influence of the ARD. Based on these reference values, the reductions in macroinvertebrate species richness observed at the two sites are 41% upstream of the mine and 85% downstream respectively.

When using total metal concentrations in the ERA, the code predicted that 84% of macroinvertebrates would be adversely affected by the measured metal concentrations immediately above the mine and 100% immediately below. Using metal bioavailability, the codes predictions were altered to 44% and 100% respectively. Thus, whilst still being ecologically conservative, it can be seen that using speciation modelling can provide a much more reasonable and realistic ERA. The ERAs were performed using ecotoxicity data for all taxa as well as just for macroinvertebrates. Similar results were achieved using both sets of data and hence macroinvertebrates seem to be a reasonable surrogate for the wider ecological community in this system.

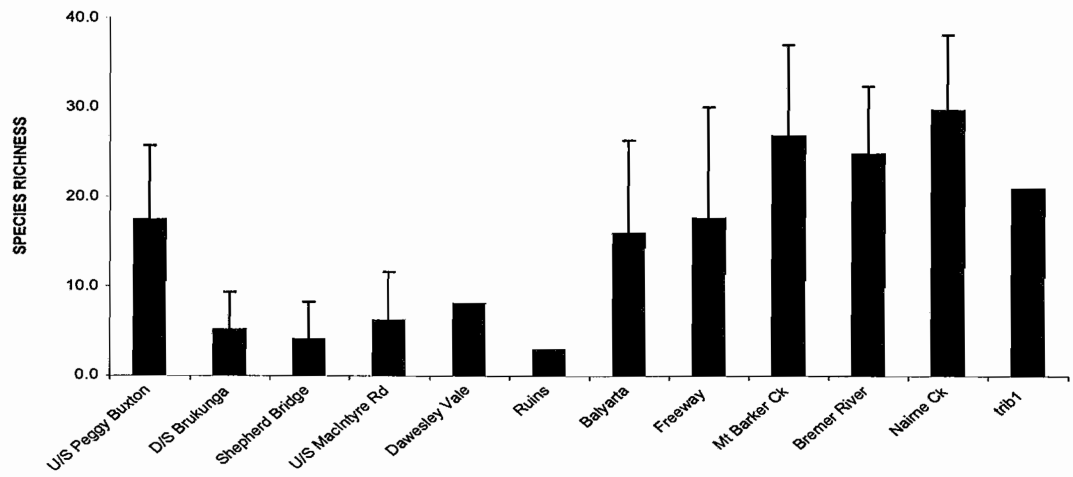

Figure 2. Results of macroinvertebrate monitoring in the Dawesley Creek/Bremer River system. The first two sites are those being evaluated in this study. The last four sites were used to provide an average reference value (see text). Species richness indicates the proportion of taxa observed at the site as a proportion of the total number of taxa observed in the system. The data are means (± s.d.) for sampling between September 1996 and December 1998.

The code was also able to estimate the degree of reduction in average metal water concentrations required to achieve acceptable impact (measured either by exceedence of criteria or by the proportion of species likely to be affected). Assuming that a recovery to 75% of species is acceptable for this highly impacted site, then the required dilution in total metal levels was approximated to be a factor of 2000. With speciation modelling, the required reduction was approximately 500. That is, the engineering target was reduced by a factor of 75%. This result implied significant benefits in relation to remedial engineering or water treatment options for affected sites. These results thus show the ability of AQUARISK to provide the environmental engineering targets, that are relevant to agreed or acceptable ecological quality measures. This link, between impact and management has been limited or missing to date.

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Current knowledge base for decision making at the Brukunga mine site

Objective for water quality

The Brukunga Mine Site Remediation Board has developed, as part of its action plan, water quality goals for water in the Dawesley Creek. The water must be suitable for domestic, stock and primary production use. Such clear goals are a crucial first step in any mine site remediation. It is unfortunately the case that such clear guidelines are frequently absent in many mine site remediation projects.

Pollution sources

Effluent from a sewage treatment plant and acid rock drainage (ARD) have been identified as the major pollution sources in Dawesley Creek in the vicinity of the mine site. The sewage treatment plant is immediately upstream of the mine site. The ARD is from waste rock dumps and a tailings dam at the mine site. The major features of the ARD are low pH (2.5–4.5) and constituents such as sulfate, aluminium and zinc at concentrations high enough to pose environmental risk to the aquatic system. As the ARD acts as a crude secondary treatment for the sewage effluent it was stressed by a number of presenters that any remediation project to improve the water quality in Dawesley Creek should take account of both pollution sources.

Acid rock drainage

Sources of ARD

The 1995 report by Environmental Geochemistry International (EGI 1995) identified three sources of ARD—the waste rock dumps, the tailings dam and the quarry together with its surrounds. Of the total sulfate load of about 3000 tonnes/year the waste rock dumps contribute about 2000 tonnes/year and the other two sources 400–500 tonnes/year each. It is estimated about 40% of the sulfate generated at the mine site is removed by the plant set up on site to treat ARD. The toes of the two largest waste rock dumps, particularly the Southern waste rock dump, are very close to Dawesley Creek. The consequence is that much of the effluent from these waste rock dumps probably enters the creek in ephemeral near surface aquifers.

Pollutant generation mechanisms

(a) Tailings dam

Bennett (1994) (incorporated in the 1995 EGI report) showed that sulfide oxidation in the tailings dam was limited to about the top 3 m. The oxygen concentrations in this 3 m region were time dependent. Preliminary analysis indicated that this time dependence could be related to the diurnal variation in atmospheric pressure. This phenomenon has been noted at other mine sites. No quantitative explanation is currently available, but it has been speculated that the phenomenon represents advective gas transport between regions of comparatively large volume connected by small volume regions of comparatively high porosity. Such a picture is consistent with that given by Taylor (2000), of layers of 'hard pan' formed a few metres below the surface of the tailings dam. Whatever the current sulfide oxidation rate, it is likely that, as in most tailings dams which have been allowed to drain, secondary mineralisation is now determining the concentration of constituents in water infiltrating the tailings dam material.

(b) Waste rock dumps

Bennett (1994) showed that, while the bulk of the material in the waste rock dumps has a very low sulfide oxidation rate, there are pods of material with oxidation rates between 6 and 250 times greater than that of the bulk rate. If all of the material in the dumps had the lower oxidation rate then the sulfate load would only be some 800 tonnes/year. The shortfall can be made up from oxidation in the pods. The fraction of the higher rate material required ranges from 0.6% to 25% of the total mass, as the pod material oxidation rate ranges from 200 to just 10 times that of the bulk rate.

(c) Factors effecting sulfate load

If secondary mineralisation is determining the concentration of many constituents of pollution in water infiltrating the tailings dam material, then reduction in the water flux through the material will reduce the load in effluent. The concentration will remain close to their present levels until dissolution of the secondary mineralisation. The material properties and geometry of the pods in the waste rock dumps impacts on the time dependence of the sulfate loads in effluent from the dumps. If much of the material has the very highest rates then we expect the load to decrease on a timescale of years. If the rate is only six times the bulk rate, then the decrease will occur over tens of years and the sulfate load will then continue at the level of about 800 tonnes/year. The cost effectiveness of any method used to reduce pollutant generation rates within the dumps requires knowledge of the size, distribution and nature of the pods. Similarly, the effectiveness of a reduction of the water flux depends critically on the pollutant generation properties of the pods.

Biological detriment

Measurements have been made on the diversity and abundance of aquatic species in the Bremer River system, from Dawesley Creek above the mine site to where the river is impacted by salinity. The species ranged from bacteria, through benthic diatoms, algae and fungi to macroinvertebrates. Organisms range in trophic level to primary producers to consumers. Their abundance and diversity can be used as a measure of the ecological health of the river system.

All measures showed a similar correlation with water quality. That is, the poorer the water quality, the lower the diversity and abundance of aquatic biota. In this sense all measures provided a consistent picture of biological detriment in the length of the river system studied, particularly that part impacted by the effluent from mine wastes at the Brukunga mine site. With some measures, the picture of biological detriment was clearer with than with others. This result was very satisfying in that the consistency ranged over such a wide trophic range. The implication is that any of the measures can, in principle, can be used to quantify biological detriment. The impact on macroinvertebrates also indicated a seasonal pattern. Although elements of this pattern repeat over the three year study there are detailed differences. The reasons for these detailed differences are not yet apparent.

Application of AQUARISK
The ANSTO code AQUARISK (Twining *et al.* 2000) was used to quantify the ecological risk to aquatic species in Dawesley Creek at the mine site. The concentration-response data used was that in the ANZECC/ARMCANZ guidelines. There was good agreement between the impact quantified using the code and field observations. In some ways this was an unexpected result given the reservations that have been expressed on the general applicability of the concentration-response data.

The result is a very reassuring, as it implies that the concentration-response data is almost certainly applicable to the present study. Moreover, the consistency of the field observations together with agreement between these field observations and an estimate of ecological risk, mean that AQURISK is a tool that can be used to quantify the changes in water quality required to improve the ecological health of the river system. Consideration should be given to using AQUARISK to define water quality goals to augment the goals currently set by the Brukunga Mine Site Remediation Board.

Future directions suggested by current knowledge base
Water quality goals

- Use AQUARISK to define water quality goals that reduces ecological risk to an acceptably low level.
- Establish that such water quality goals are consistent with those currently set by the Brukunga Mine Site Remediation Board.
- Recommend to the Brukunga Mine Site Remediation Board any augmentation or modification of their current water quality goals.

Ecological detriment

- Examine the different measures of ecological detriment and develop such criteria as ease of sample collection, ease of data analysis and time lapse between data collection which can be used to assess the relative cost effectiveness of the different measures.
- Recommend which measures to use to assess the effectiveness of remediation implemented at Brukunga mine site.
- Assess the extent to which techniques and measures used in the present study can be applied to other Australian rivers systems.

Mine site

- Establish that the water balance data at the mine site is adequate to assess both the effectiveness of proposed remediation and remediation schemes once implemented.

- Establish that the mass balance for the major constituents of pollution is adequate to assess both the effectiveness of proposed remediation and remediation schemes once implemented.
- Develop a model of water flow and mass load in the river system covering the region from immediately above the mine site to where the Bremer River is impacted by salinity. This model should of sufficient precision to assess the effectiveness of proposed remediation and remediation schemes once implemented.

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SYNOPSIS

The Third Brukunga Workshop reported on progress since the last workshop (November 1998), and where possible, summarised the current state of knowledge of the effects of acid rock drainage (ARD) from the Brukunga mine site on downstream aquatic ecosystems in the Dawesley Creek/Bremer River system. The workshop was organised by ANSTO to facilitate the achievement of objectives within the Managing Mine Wastes Project relating to biological measurement of the impacts of ARD on aquatic ecosystems. More generally, the workshop provided both a venue to report on studies of the Brukunga site—an ideal 'natural laboratory', and a forum at the local level where information could contribute to the development of a strategy to remediate the aquatic ecosystems of the Dawesley Creek/Bremer River system to an acceptable level, following a risk-based approach.

Presentations were given by a diverse range of stakeholders. Hume MacDonald, chairman of the newly established Brukunga Mine Site Remediation Board, outlined the Board's terms of reference and details of work being undertaken by independent bodies to provide solutions to remediating the mine site and its environs. It was generally agreed that the new Board's vision of achieving practical outcomes through a series of ongoing, focus-driven projects was commendable. Ivan Dainis proposed a conceptual model to best explain the seasonal behaviour of the load of acid sulfates and associated metals that impact on Dawesley Creek as a result of the Brukunga mine. The model is based on flow proportional monitoring in 1998 and earlier historical data. It explains the variable Al and Fe values observed and also provides tools for the identification and explanation of anomalous monitoring results. This new understanding of elemental variations and inter-relationships should simplify monitoring and assist the inter-correlation of biological testing results with those from chemical monitoring. Graham Taylor presented a summary of work conducted by students, sponsored by the CSIRO Minesite Rehabilitation Research Program, relating to understanding acid mine drainage impacts and improving remedial strategies at Brukunga.

Several presentations were given on the use of biological indicators of water quality in the Dawesley Creek/Bremer River system. Work with microorganisms (algae and fungi) was greatly expanded over the past year. Karyn Wilde and coworkers assayed the functional activity of microbial communities upstream and downstream of the Brukunga mine by measuring community level physiological profiles using BIOLOG microwell plates. Multivariate statistical analyses successfully distinguished between ARD polluted, salinity affected and reference sites suffering varying degrees of eutrophication. Peter Holden and coworkers used phospholipid-fatty acid (PLFA) analysis to quantify the effect of acid mine drainage on the diversity and abundance of sediment microorganisms upstream and downstream of the mine. In support of Karyn Wilde's work, PLFA analysis showed low microbial activity at ARD affected sites compared to reference sites, and appeared to correlate with measured water quality. This result was also confirmed using aminopeptidase activity and dimethyl sulfoxide reductase assays. Rebecca Edwards investigated the use of fungi as an indicator of ARD. A definite shift in fungal community structure was observed at the sites affected by ARD. Laboratory studies demonstrated a shift-back in fungal community structure when fungi isolated downstream of the mine were placed in 'reference' water, indicating recovery potential.

Adam Sincock and coworkers examined the community structure of benthic microscopic algae (diatoms) as an indicator of water quality, particularly the impact of ARD, in the Dawesley Creek/Bremer River system. Like the studies reported by Karyn Wilde and Peter Holden, multivariate statistical analyses successfully distinguished between ARD polluted, salinity affected and reference sites suffering varying degrees of eutrophication. The degree of pollution was related to the flow in Dawesley Creek (see below). John Ferris provided a summary of algal studies at Brukunga over the past three years, including the field-based benthic diatom work and the laboratory-based fluorescein diacetate ('metabolic probe') work. The latter work assessed the short-term physiological responses of a unicellular Green and a Blue-green alga to water sampled from Dawesley Creek/Bremer River. In general, metabolic activity increased when exposed to sewage effluent and decreased when exposed to ARD. The laboratory based bioassay was also able to distinguish the three major pollutants in the Dawesley Creek/Bremer River system. John Ferris also compared their diatom field studies with data for macroinvertebrates and fish and concluded that broadly similar results were obtained in the system. Similarly, Peter Holden reported that results for PLFA were consistent with those for macroinvertebrates.

Peter Schultz summarised the three years data on the diversity and abundance of macroinvertebrates in the Dawesley Creek/Bremer River system with emphasis on the impact of ARD. In general, it was concluded that water in Dawesley Creek is highly acidic and unsuitable for macroinvertebrates. The downstream extent of the impact reaches the to the confluence of Dawesley and Nairne Creeks at all times, except when extreme flood discharges dilute the system for brief periods. Impact between this confluence and Mount Barker Creek is dependent on stream flows (see below). John Twining reported on further development of AQUARISK, a computer program developed for ecological risk assessment. It was applied in a case example to evaluate the likelihood of biotic (macroinvertebrate) detriment, due to ARD, at a site upstream and downstream of the mine. In short, once metal bioavailability was taken into account, the predictions of AQUARISK were close to those measured independently from the field. AQUARISK was also able to estimate the degree of reduction in average metal water concentrations required to achieve acceptable impact (measured either by exceeding guidelines or by the proportion of species likely to be affected).

Ian Ritchie provided a very useful summary of our current knowledge base at Brukunga and perceived future directions.

Based on ecological data collected over the past three years a consistent pattern of impact on the Dawesley Creek/Bremer River system from the Brukunga mine is evident. That is, the diversity and abundance of biota is typically very low in Dawesley Creek up to 25 km downstream of the Brukunga mine (*i.e.* to the confluence with Mt Barker Creek). This is, however, moderated by stream flows. When flow is higher in Dawesley Creek, relative to Nairne Creek (17 km downstream), the effects of AMD are evident downstream to the Mount Barker confluence (usually in Spring). However, when flow is higher in Nairne Creek, relative to Dawesley Creek, this ameliorates the pollution gradient, and hence, considerably improves diversity and abundance of some organisms downstream to the Mount Barker confluence. Surface water flow is a dynamic phenomenon in the system and deserves a more detailed investigation. Another important aspect of the ecological work is that similar patterns of ARD impact are expressed using a range of organisms (from microorganisms to fish) and endpoints (metabolic probes to community structure). More importantly, the use of multivariate statistical analyses has permitted ecological impact from ARD, nutrient eutrophication and salinity to be separated.

It is apparent that field survey data are a fundamental requirement to enable adequate description of the degree of ecological detriment downstream of Brukunga, as well as to indicate the scope for recovery within an environment that is otherwise highly modified by human activity. These measures, however, have been nicely complemented using a suite of laboratory-based 'physiological' measures, which lend more scope to understanding the mechanisms of impact. Some of the microbial measures offer a relatively cheap and rapid method of quantifying impact of ARD on rivers and streams. There is clearly a move towards measuring the ecological effects of pollutants, such as AMD, rather than relying simply on physico-chemical water quality data. Further, we are developing a suite of biomonitoring tools from which to select the one(s) appropriate to particular situations, rather than attempting to define a 'panacea' group to be applied in all circumstances. The current biological data, together with the suite of biomonitoring tools, also provides the means by which the success of any remediation strategy can be measured and monitored over time.

The risk assessment program, AQUARISK, was further developed over the past year. Local water quality and biological effects (based on macroinvertebrates only) data were integrated at two sites in Dawesley Creek (upstream and downstream of the mine) to determine the likelihood of ecological detriment due to ARD. It was encouraging that the ecological effects predicted by the model were in close agreement with the measured data. This 'verification', at least for macroinvertebrates, provides confidence in the model. Macroinvertebrates appear to be a reasonable surrogate for the wider ecological community in the Dawesley Creek/Bremer River system. AQUARISK was also able to estimate the degree of reduction in average metal water concentrations required to achieve acceptable impact (measured either by allowable exceedence of criteria or by the proportion of species likely to be affected). Assuming that a recovery to 75% of species is acceptable for this highly impacted site, then the required dilution in total metal levels was approximated to be a factor of 2000. With speciation modelling, the required reduction was approximately 500. That is, the engineering target was reduced by a factor of 75%. This result implied significant benefits in relation to remedial engineering or water treatment options for affected sites. These results thus show the ability of AQUARISK to provide the environmental engineering targets, that are relevant to agreed or acceptable ecological quality measures. This link, between impact and management has been limited or missing to date.

Outcomes from the workshop include these proceedings, which can be used as a reference for those who may be interested in the research direction being applied to assess the current level of impact at the site. This information in turn may be used to assist in the derivation of appropriate, environmentally relevant, rehabilitation criteria. In addition, a contact network of interested parties was extended. Given adequate communication, this will help to ensure an efficient use of limited research resources being applied to a range of approved studies. It should also help to ensure that appropriate and cost-effective management strategies for the Brukunga mine site are implemented.

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