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Diatoms and macroinvertebrates as biomonitors of mine-lakes in Collie, Western Australia

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Abstract

Several voids created through open-cut coal mining occur near the town of Collie in the south-west of Western Australia. After mining, the voids fill with fresh water and form mostly acidic wetlands. Five of these mine-lakes were monitored in 1999 using macroinvertebrates and diatoms. On the basis of acidity and water chemistry two groups of wetlands were identified using multivariate analyses; wetlands with low pH (< 4.5), and those with comparatively higher pH (> 4.8). Distinct macroinvertebrate and diatom assemblages were characteristic of each of the wetland groups. Macroinvertebrates including *Orthetrum caledonicum* and *Megaporus solidus* were associated with the Group 1 wetlands (pH < 4.5) while *Sternopriscus browni* and *Micronecta* sp. were two of the most abundant macroinvertebrates in the Group 2 wetlands (pH > 4.8). In the Group 1 wetlands *Nitzschia paleaeformis* and *Pinnularia microstauron* were among the dominant diatom species. *Eunotia curvata* and *Tabellaria flocculosa* were two of the diatom species commonly found in the Group 2 waterbodies. While pH was one of the factors primarily responsible for the distribution of both biomonitors, diatoms appeared to be more sensitive than macroinvertebrates to acidity.

Keywords: Diatoms, macroinvertebrates, pH, mine-void wetlands, biomonitors

Introduction

The continuing loss and degradation of the world's natural wetlands has triggered an increased emphasis on artificial wetlands (Hammer 1992). Pits, or voids, created by mining processes may be rehabilitated to resemble natural habitat or to be used for recreational purposes (John 1993). The Collie Basin, a depression in the Yilgarn Block in South-western Australia has many voids created after open-cut coal mining (Lord 1979) that intercept the aquifer and eventually become wetlands (Brugam & Lusk 1986). The water from artificial wetlands created by mining can be problematic with high acidity and extreme concentrations of heavy metals (John 1997; Kalin *et al.* 2001). The mine-void wetlands of the Collie Basin range in pH from 2–6. Sources of acidity in mine voids can include the unearthing of sulphidic soil (Gambs & Walsh 1981), leaching of mine waste dumps and erosion (Norris 1986). However, the exact cause of acidity in the Collie voids is uncertain although sulphidic soil and organic acids are implicated (John 1997).

Only limited work has been carried out on the biological communities of the Collie void wetlands. Ewington 2, Blue Waters, Stockton Lake, Stockton Tailings Pond and Black Diamond were the five wetlands chosen for this study. Two commonly used biomonitors, macroinvertebrates (Abel 1989; Hellawell 1986; Lenat & Barbour 1994; Rosenberg & Resh 1993) and diatoms (Battarbee *et al.* 1999; Chessman *et al.* 1999; Lepistö 1988) were selected as potential ecological indicators for these wetlands. Biological monitoring is commonly used as chemical variables lack the responsiveness necessary to assess ecosystem health (John 2003). Chemicals in aquatic systems can undergo wide fluctuations and while chemical monitoring can provide information about the conditions at the time of sampling, it may not detect biologically significant peak concentrations (Abel 1989). Whereas biological monitors such as macroinvertebrates integrate changes in water quality (Sandin, Dahl & Johnson 2004).

The objectives were to identify the macroinvertebrate and diatom assemblages characteristic of the wetlands and to determine species distribution along a pH gradient. A further objective of this study was to determine which group was the most sensitive indicator of acidity.

Analyses of both invertebrate (Courtney & Clements 1998; McNicol *et al.* 1995; Schindler 1990) and diatom communities (Stokes & Yung 1986; Eloranta 1990; Kwandrans 1993; Battarbee *et al.* 1997) have documented shifts in community structure and reduced diversity in response to low pH. Although macroinvertebrates are more frequently used as biomonitors in Australia, studies have shown that diatoms are highly sensitive to water quality changes (John 1993).

Monitoring biodiversity in acid–impacted water bodies has become an integral part of the management strategy for such waters. The use of biomonitors such as macroinvertebrates and diatoms can provide greater understanding of these systems, from the perspective of rehabilitation. Development of ideal tools for biomonitoring the success of converting acidic voids into functional wetlands may have significant uses throughout Australia.

Methods

Study Site

The town of Collie (32°S, 116°E) is approximately 200km south of Perth, Western Australia. The study sites (Fig 1, Table 1) are located south-east of the town in the Collie Basin, which covers an area of about 274km^2 (Environmental Protection Authority 1992).

Black Diamond is a large void wetland with boundaries comprising of low gradient shoreline and sections of steep cliff. It possesses a moderate level of riparian vegetation. Blue Waters has similar cliffs to Black Diamond but is a smaller wetland, basically devoid of fringing vegetation. Ewington 2 is relatively small but has a maximum depth of 11 metres. There is a dense stand of sedges growing at the north-east corner of the void. Stockton Lake is one of the larger wetlands sampled with a long cliff face along one side. There are some emergent macrophytes present and it is the only void from which the native fish *Edelia vittata* has been collected. Stockton Tailings Pond is a small wetland with abundant fallen leaves and debris from the peripheral vegetation and a high deposition of iron oxide. It is the only lake out of the five sampled to display extensive macroscopic algal growth. *Mougeotia* sp., a filamentous green alga recognised as a hyperaccumulator of iron and aluminium (John *et al.* 1999), was recorded in abundance in this lake.

Table 1

Code and location of void wetlands

Wetland Name	Wetland Code	GPS
Black Diamond	BD	33°20.33s 116°05.58e
Blue Waters	B	33°20.24s 116°13.16e
Ewington 2	E	33°20.48s 116°12.02e
Stockton Lake	S	33°23.13s 116°13.°75e
Stockton Tailings Pond	ST	33°23.13s 116°13.74e

Sampling

The sites were monitored monthly between March and July 1999. Electrical conductivity, salinity, pH, temperature and dissolved oxygen were measured using a T.P.S WP-81 Meter and a portable HORIBA Water Quality Checker (U – 10).

Unfiltered water samples were collected from each wetland using acid treated bottles, and were analysed by

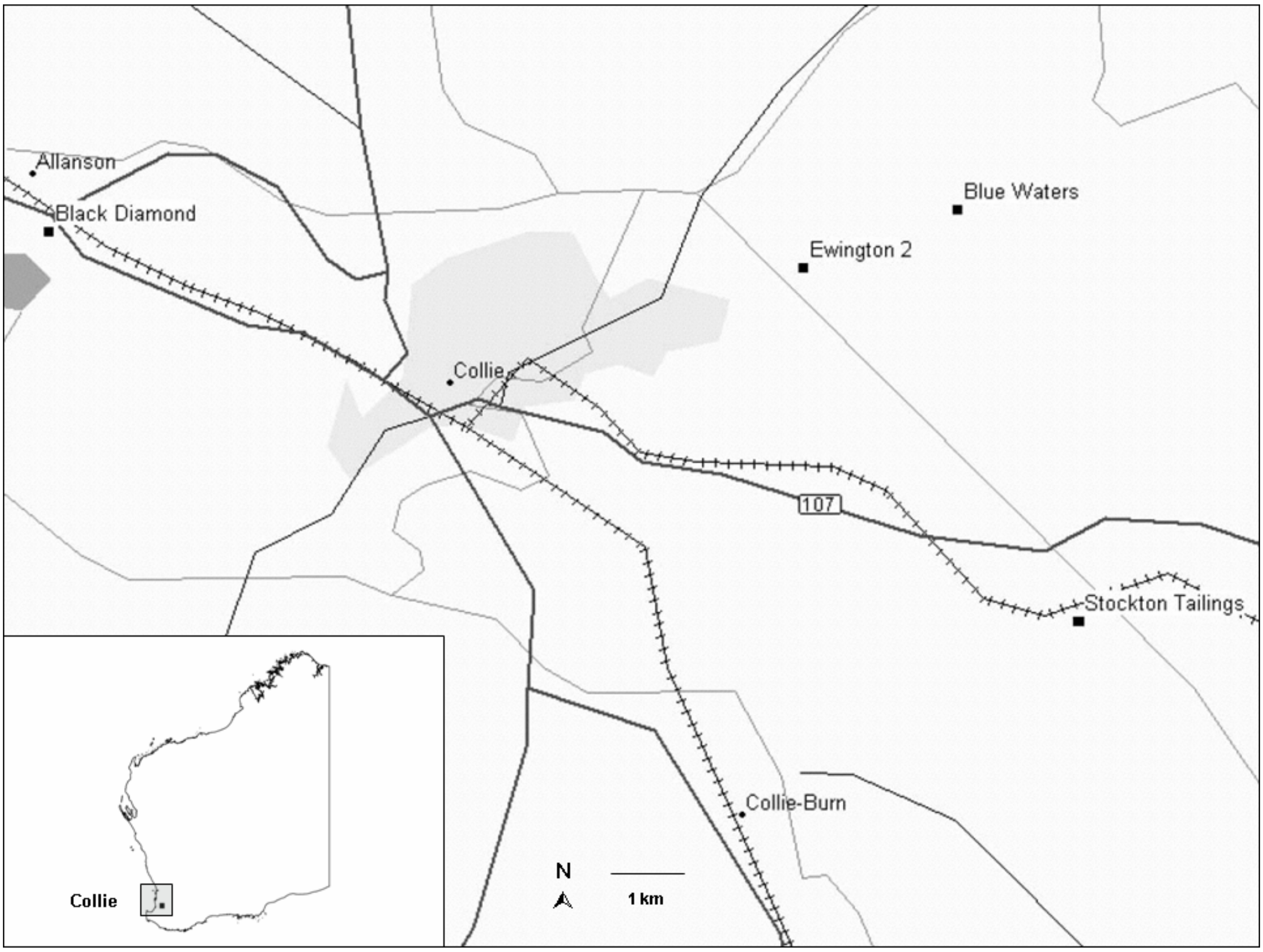

Figure 1. Map showing location of the Collie wetlands.

the Australian Environmental Laboratories (Analabs) Pty Ltd., Welshpool, Western Australia for pH, total alkalinity, calcium, aluminium, iron (soluble), total carbon, total organic carbon, total nitrogen, total phosphorus and chlorophyll *α*.

Macroinvertebrate and diatom samples were collected fortnightly to monthly. Macroinvertebrates were monitored using a rapid assessment method modified from Chessman 1995. Samples were collected using a 250 μm mesh net with a 400 mm x 280 mm opening. At least three habitats including emergent vegetation and benthos were sampled at each wetland, over a 6 metre transect. The collected material was sieved and specimens live picked for a minimum of 30 minutes, with samples preserved in 10 % formalin. Periphytic diatoms were collected using JJ Periphytometers – to ensure uniform sampling (John 1998). These were fitted with ten glass microscope slides and were fully immersed at the sites. After at least fourteen days, they were retrieved and the slides (with the periphyton) were removed and preserved using Transeau's algal preservative.

Laboratory Methods

Macroinvertebrates were identified to species where possible using specialised literature: Williams (1980), Davis & Christidis (1997), Horwitz (1995), Ingram *et al.* (1997) and Hawking & Smith (1997). All macroinvertebrates present in the samples were counted and the frequency of the different taxa recorded for statistical analyses. The dominant species were photographed using an Olympus Zoom Stereo-microscope and Olympus SC35 Camera. Voucher specimens were deposited in the Department of Environmental Biology, Curtin University of Technology.

The periphytic samples were processed into permanent slides following methods outlined in John (1983). Between 100 and 350 diatom valves were counted for each sample and percentage frequencies calculated for each species. Light micrographs were taken using a Vanox photomicroscope. Identification was carried out using specialised literature (Patrick & Reimer 1966; Foged 1978; John 1983, 1993, 1998). The permanent slides were deposited in the International Diatom Herbarium, School of Environmental Biology, Curtin University of Technology.

Data Analysis

Multivariate analyses were used to classify the wetlands based on environmental parameters and species composition. Ordinations were generated using the statistical package PC-ORD 4. Correlation based principal component analysis (PCA) was conducted to group sites according to environmental variables, while canonical correspondence analysis (CCA) was used to relate species composition to environmental variables. The statistical significance of variables was established using the Monte Carlo permutation test with 999 random permutations. Species abundance data and the water quality parameters of salinity and temperature were $\log_{10}$ transformed prior to statistical analysis.

Univariate analyses were conducted using the statistical package Minitab version 11. Prior to analysis, data was tested using Levene's test for homogeneity of variance and the Kolmogorov-Smirnov test for normality. One-way Analysis of Variance (ANOVA) was employed to test for significant differences between the wetland groups derived from the ordinations, based on species richness and pH.

Results

As shown in Table 2, alkalinity (represented as $CaCO_3$) was low at each of the wetlands with readings of < 5 mgL^{-1}. Stockton Lake and Black Diamond Lake had substantially lower concentrations of aluminium and iron and Stockton Tailings Pond had the highest levels of iron while aluminium was highest at Ewington 2. TN and TP were low in all wetlands. Productivity (as indicated by chlorophyll *a* concentration) was also low.

All five wetlands were classified as acidic (Table 3). Although the highest salinity and electrical conductivity readings were approximately four times higher than the lowest, all five wetlands were fresh (salinity < 3 ppt). Seasonal differences meant that both temperature and dissolved oxygen displayed a large variation over the sampling period (Table 3).

The environmental data were subjected to principal components analysis to determine the inter-relationship of the wetlands. Ordination resulted in the separation of the sites into two groups, defined by the first two principal components (Fig 2). A large proportion of variance along axis 1 was explained by the variable pH (r = -0.874). Axis 2 was primarily influenced by salinity (r = 0.878).

There was a significant difference between the pH of the two groups according to ANOVA results ($P < 0.001$, F = 170.89, df = 52). The first group of wetlands possessed a

Table 2

Water chemistry of the five Collie wetlands (Ca – calcium, Al – aluminium, Fe – iron (soluble), TC – total carbon, TOC – total organic carbon, TN – total persulphate nitrogen, TP – total persulphate phosphorous and Chl *α* – chlorophyll *α*). Measured in mgL^{-1}

Site	Alkalinity (as $CaCO_3$)	Ca	Al	Fe	TC	TOC	TN	TP	Chl *α*
Black Diamond	< 5	6.5	0.2	< 0.05	4.1	3.6	0.21	< 0.01	0.0012
Blue Waters	< 5	10	3.4	0.6	2.8	2.4	0.14	< 0.01	0.0049
Ewington 2	< 5	6.4	4.2	1.9	5.2	3.4	0.18	0.04	0.0009
Stockton Lake	< 5	3.6	0.4	0.5	4.5	4.2	1.1	< 0.01	0.0008
Stockton Tailings	< 5	12	2	6.7	2.2	1.9	0.7	< 0.01	0.0025

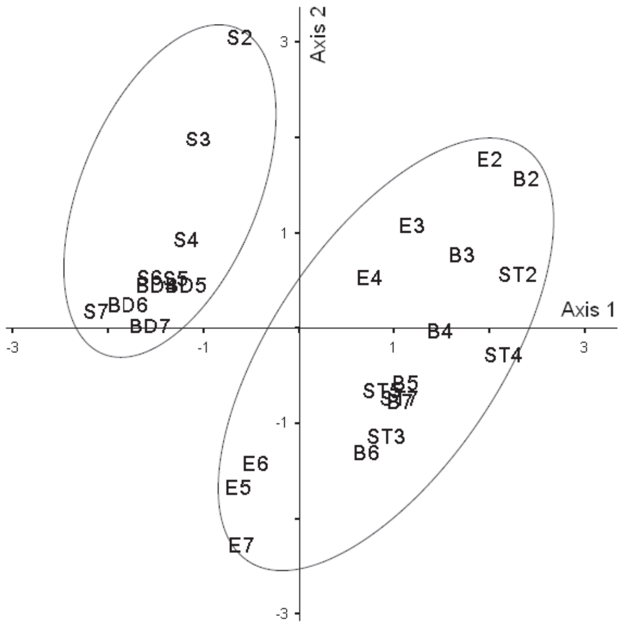

Figure 2. Principal component analysis (PCA) of the Collie wetlands based on environmental variables monitored over the sampling period (March – July). The cluster to the right of the ordination (Group 1) consists of Ewington 2, Stockton Tailings Pond and Blue Waters and represents sites with lower pH values (pH < 4.5). The cluster to top left (Group 2) includes Stockton Lake and Black Diamond Lake and represents sites with pH > 4.8. B – Blue Waters; BD – Black Diamond; E – Ewington 2; S – Stockton Lake; ST – Stockton Tailings Pond. The numbers 2–7 indicate the sampling occasion.

mean pH of 3.96 ± 0.39 while the second group possessed a mean of 5.29 ± 0.26.

The differences in the distribution of invertebrate taxa in the wetlands were determined by canonical correspondence analysis (CCA). Axes 1 and 2 of the CCA explained 18.5% of the total variance in macroinvertebrate distribution (Fig 3). The eigenvalues of axes 1 and 2 were both significant (P < 0.01) with values of 0.385 and 0.220 respectively. There was a strong relationship between macroinvertebrate community structure and water quality parameters that was supported by the high species-environment correlations of axis 1 (r = 0.935) and axis 2 (r = 0.837). Axis 1 was primarily related to pH and salinity, separating the wetlands into two main groups. Group 1 comprised of wetlands with acidic pH (< 4.5) while the Group 2 wetlands possessed comparatively higher pH levels (> 4.8) and lower salinity concentrations.

Although differences were detected in the macroinvertebrate assemblages present, the ANOVA determined that there was no significant difference between the species richness of the two groups (P > 0.05, F = 1.21, df = 25), with mean values of 7.67 ± 3.12 and 8.59 ± 3.28 respectively.

Invertebrate taxa that dominated the wetland groups are listed in Table 4. Eight invertebrate taxa were identified as dominant in Group 1 wetlands, with adult coleopterans accounting for the largest proportion. Coleopterans also dominated the Group 2 wetlands with three of the five most commonly occurring invertebrates belonging to this order. Some overlap of dominant taxa was evident between the two groups with the hemipteran *Micronecta* sp., the coleopteran *Necterosoma darwini* and the crustacean *Perthia* sp. inhabiting both wetland types.

Canonical correspondence analysis of the diatom data showed that 35.3% of the total variance in community

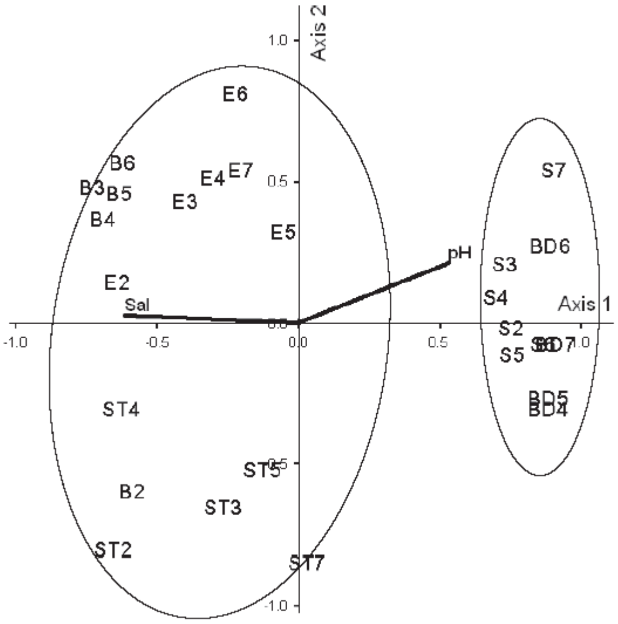

Figure 3. CCA biplot showing wetlands and water quality parameters. The ordination is based on sites, the parameters of pH, dissolved oxygen, $\log_{10}$ salinity, $\log_{10}$ temperature and $\log_{10}$ macroinvertebrate abundance data. Bold lines represent primary water quality parameters separating the two groups. B – Blue Waters; BD – Black Diamond; E – Ewington 2; S – Stockton Lake; ST – Stockton Tailings Pond. The numbers 2–7 indicate the sampling occasion.

Table 3

The range of water quality parameters from Collie wetlands sampled. EC – Electrical Conductivity; DO – Dissolved Oxygen. Numbers of sampling occasions: Black Diamond – 4, Blue Waters – 6, Ewington 2 – 6, Stockton Lake – 6, Stockton Tailings Pond – 5

Site	pH	EC (mS/cm)	Salinity (ppt)	Temp (°C)	DO (ppm)
Black Diamond	4.89 – 5.38	385 – 422	0.21 – 0.23	12.6 – 16.2	7.15 – 8.10
Blue Waters	3.43 – 4.14	1180 – 1931	0.66 – 0.94	12.9 – 22.6	7.28 – 9.01
Ewington 2	3.79 – 4.38	937 – 1465	0.51 – 0.65	12 – 22.1	6.22 – 9.25
Stockton Lake	5.01 – 5.93	393 – 487	0.20 – 0.25	12.5 – 21.4	7.23 – 8.99
Stockton Tailings	3.11 – 3.5	858 – 1285	0.51 – 0.70	12 – 22.9	7.05 – 8.87

Table 4

List of invertebrate taxa recorded from the two groups of wetlands (taxa included were present at > 20% frequency). a indicates adult Coleoptera.

Group 1 Taxa (pH < 4.5)	Group 2 Taxa (pH > 4.8)
Anisops sp.	*Micronecta* sp.
Diplacodes bipunctata (Burmeister)	*Necterosoma darwini* (Babington) **a**
Megaporus howitti (Clark) **a**	*Perthia* sp.
Megaporus solidus (Sharp) **a**	*Sternopriscus browni* (Sharp) **a**
Micronecta sp.	*Sternopriscus maedfooti* (Clark) **a**
Necterosoma darwini (Babington) **a**	
Orthetrum caledonicum	
Perthia sp.	

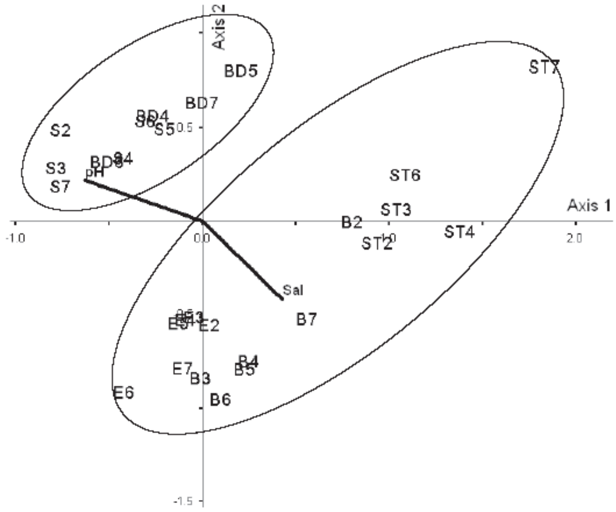

Figure 4. CCA biplot showing wetlands and water quality parameters. The ordination is based on sites, the parameters of pH, dissolved oxygen, $\log_{10}$ salinity, $\log_{10}$ temperature and $\log_{10}$ diatom abundance data. Diatom species included were those with a maximum abundance of > 1%. Bold lines represent primary water quality parameters separating the two groups. B – Blue Waters; BD – Black Diamond; E – Ewington 2; S – Stockton Lake; ST – Stockton Tailings Pond. The numbers 2–7 indicate the sampling occasion.

structure was explained by the first 2 axes (Fig 4). With eigenvalues of 0.475 and 0.297 respectively, both axis 1 and axis 2 were significant ($P < 0.01$). High species-environment correlations for axis 1 ($r = 0.893$) and axis 2 ($r = 0.887$) displayed the strong relationship between diatoms and water quality parameters, similar to that demonstrated by macroinvertebrates (Fig 3). pH and salinity were the two factors most closely related to axis 1 and were the primary factors determining the separation of the wetlands into two groups (Fig 4).

Diatom species richness varied significantly between the two groups. ANOVA determined that the mean taxa numbers of the Group 1 wetlands (8.46 ± 2.60) were significantly lower ($P < 0.001$, F = 109.29, df = 52) than the Group 2 wetlands (16.39 ± 2.64). Six species were identified as the most commonly occurring taxa in the first group while eight species dominated the diatom flora of Group 2 (Table 5). There was a limited amount of overlap between the two groups as *Brachysira brebissonii* and *Achnanthidium oblongella* were abundant in both the Group 1 and Group 2 wetlands.

Discussion

Water Quality

Waters affected by mining processes are often highly acidic (Brugam & Lusk 1986; John 1997) consistent with the pH (< 4.5) of Ewington 2, Blue Waters and Stockton Tailings Pond (Group 1). Although the pH of Stockton Lake and Black Diamond (Group 2) were relatively low they were comparatively higher than the Group 1 wetlands (pH > 4.8). Stockton Lake and Black Diamond were further differentiated from the other wetlands by lower salinity, which in conjunction with pH contributed to the separation of the wetlands into two groups in the multivariate analyses. Lower concentrations of inorganic ions were another distinguishing feature of the Group 2 wetlands. Acidification is related to elevated levels of certain metals (Lydén & Grahn 1985) with high concentrations of inorganic ions including calcium, magnesium, sulphate and aluminium common to waters

Table 5

Dominant diatom taxa recorded from the wetland groups (taxa included were present at > 20% frequency).

Group 1 Taxa (pH < 4.5)	Group 2 Taxa (pH > 4.8)
Achnanthidium oblongella (Oestrup)	*Achnanthidium oblongella* (Oestrup)
Brachysira brebissonii (Ross)	*Brachysira brebissonii* (Ross)
Epithemia sorex (Kütz.)	*Eunotia curvata* (Kütz.) Lagerst
Navicula aff. *cari* (Ehr.)	*Eunotia exigua* (de Breb.) Grun.
Nitzschia paleaeformis (Hust.)	*Frustulia* sp.
Pinnularia microstauron (Ehr.)Cl.	*Rhopalodia gibberula* (Ehr.) O. Müller
	Surirella tenera Greg.
	Tabellaria flocculosa (Rabh.) Kütz.

affected by mining (John 1997). The lower concentrations of inorganic ions at the Group 2 lakes as opposed to Group 1 support these findings. The reasons for the differences in water chemistry may be related to age of the wetlands. Black Diamond and Stockton Lake were terminated as mines in 1953 and 1957 respectively and subsequently filled with fresh water (Stronach 1988) and have naturally undergone neutralisation to some degree. Blue Waters and Ewington 2 remained operational for several years after the closure of Black Diamond and Stockton Lake (Ashton 1988).

Macroinvertebrates

The majority of the eight invertebrate taxa identified as dominant in Group 1 (pH < 4.5) belonged to the order Coleoptera. Hemipterans, anisopterans and amphipods were also common. Although the abundance of these taxa in the Group 1 voids might imply that they would be useful indicator species, labelling them as characteristic of low pH waters would be premature. Firstly, there was an overlap between the dominant taxa of the Group 1 and Group 2 (pH > 4.8) waterbodies. The adult coleopteran *Necterosoma darwini* (a) was abundant in both wetland types while the amphipod *Perthia* sp. and the hemipteran *Micronecta* sp. followed a similar pattern of distribution.

Secondly, none of the dominant taxa are restricted to acidic waters. *Megaporus solidus, Megaporus howitti* and *Necterosoma darwini,* the three adult coleopterans that commonly occurred in the Group 1 voids, have been found in wetlands of the Swan Coastal Plain over a range of pH (Balla & Davis 1993; Davis & Christidis 1997). For example, Davis & Christidis (1997) recorded *Megaporus howitti* from Lake Jandabup and North Lake. pH levels of between 4 and 5 have been documented at Lake Jandabup (Sommer & Horwitz 2001) while North Lake is generally considered alkaline (Davis & Rolls 1987).

The crustacean *Perthia* sp. was another of the dominant taxa in the Collie wetlands. Cheal *et al.* (1993) noted that the two recognised species of *Perthia* were only in coloured wetlands. The high levels of humic and fulvic acids in highly coloured wetlands are known to contribute to lower pH (Schmidt & Rosich 1993). The findings of that study in conjunction with those of the current project indicate that *Perthia* may be tolerant to low pH. However the occurrence of *Perthia acutitelson* in Lake Yangebup (Williams & Barnard 1988), a lake identified as having high alkalinity by Growns *et al.* 1993, shows that the taxon may not be reflect acidic conditions.

Micronecta sp. was also abundant in the Collie voids. However, the use of this taxon as an indicator of acidity would be misleading as previous studies have recorded species of this genus from many wetlands in south-western Australia. While *Micronecta robusta* was reported from lakes of the Perth region: Thomsons Lake, Lake Joondalup, Lake Monger, North Lake and the acidic Lake Jandabup, they were most common in the alkaline waterbodies of Lake Monger and North Lake (Davis & Rolls 1987).

Five taxa dominated the invertebrate assemblages of the Group 2 lakes with adult coleopterans and hemipterans comprising the abundant taxa. There was a similar situation with the dominant invertebrates of the Group 2 waterbodies. None of the taxa abundant in the Group 2 lakes are restricted to those pH levels and their use as indicator species would be questionable. For example, *Necterosoma darwini* has been collected from acidic wetlands such as Lake Gnangara (Growns *et al.* 1993) through to relatively neutral lakes including Nowergup Lake and Bartram Swamp (Balla & Davis 1993). Furthermore this coleopteran was also commonly found in the highly acidic Group 1 wetlands of the present study.

The invertebrate taxa abundant in the Collie wetlands are probably opportunistic in nature. The voids are generally lacking in predators and some have stands of peripheral vegetation, two factors which may provide incentive for invertebrate inhabitants. Therefore, while the invertebrates present may be tolerant rather than indicative of the water quality conditions, they do provide insight into the functionality of the wetlands.

Diatoms

Brachysira brebissonii, Nitzschia paleaeformis and *Pinnularia microstauron* were among the most abundant diatom species in the Group 1 wetlands. *Brachysira brebissonii* was noted by Foged (1978) as acidophilous; acidophilous being species which generally occur at pH < 7 but may be found in waters of around pH 7 (Hustedt 1937–1939). John (1993) documented *Nitzschia paleaeformis* in acidic sand mining voids at Capel, Western Australia while Watanabe & Asai (2004) recorded the species from acidic waters in Japan. *Pinnularia microstauron* was identified from waters of pH 5–6 during work on east African diatoms by Gasse (1986). This species seems to prefer slightly acidic water although it can tolerate a range of pH (Patrick & Reimer 1966). The preference of these diatom species for low pH conditions is supported by their dominance in the Group 1 voids of Collie.

Although most dominant species in the Group 1 wetlands were acidophilous, the remaining two species *Achnanthidium oblongella* and *Epithemia sorex* favour alkaline conditions (Foged 1979). *Epithemia sorex* is also known to prefer waters of high conductivity (Patrick and Reimer 1975). Given the low pH of the wetlands, this suggests that the presence of the species in the Group 1 wetlands is probably due to the higher salinity/conductivity of the group in comparison with Group 2. *Achnanthidium oblongella* was described as alkaliphilous by Foged (1978 & 1979); alkaliphilous being species which may occur at pH 7 but generally occur at pH > 7 (Hustedt 1937–1939). However a study of diatoms in New Zealand showed that 11 of 45 locations from which *Achnanthidium oblongella* was collected had pH levels of between 4.5 and 6.4 (Foged 1979). These findings along with those of the current project suggest that while the species may prefer alkaline waterbodies it has a reasonable tolerance to lower pH.

Although most of the dominant diatom taxa of Stockton Lake and Black Diamond (Group 2) are acidophilous, there is a higher proportion of the abundant alkaliphilous species in comparison to the Group 1 wetlands. *Achnanthidium oblongella, Rhopalodia gibberula* and *Surirella tenera* have all been described as alkaliphilous (Foged 1978). The tolerance of

Achnanthidium oblongella to different pH levels was established earlier and presents a possible explanation for its dominance in both wetland types. *Rhopalodia gibberula* was described by Patrick and Reimer (1975) as one that appeared to prefer water with some chloride. Given that the Group 2 wetlands possessed relatively low salinity and conductivity the abundance of *Rhopalodia gibberula* seems incongruous, but Patrick and Reimer (1975) also noted that the species may be found in waters of low conductivity and is widely tolerant. The dominant acididophilous species included *Tabellaria flocculosa* (DeNicola 1986 and Ford 1986) and *Eunotia exigua* (Renberg *et al.* 1985; Ford 1986) and *Eunotia curvata* (Patrick & Reimer 1966).

There was a limited overlap of dominant diatom species between the two groups of wetlands. The acidophilous *Brachysira brebissonii* was abundant in both wetland types while the tolerant *Achnanthidium oblongella* was also dominant in both groups.

Species Richness

Macroinvertebrate taxonomic richness is progressively reduced with declining pH and as acid sensitive species are lost they are replaced by a few acid tolerant taxa (McNicol *et al.* 1995; New 1995). For example Courtney and Clements (1998) identified significantly fewer taxa at pH 4.0 than in waters with pH 5.5 and 6.5. In the present study, there was no pattern of species reduction with declining pH. In contrast, the numbers of diatom taxa in the two groups of wetlands supported findings reported in the literature (Haworth *et al.* 1990; DeNicola 2000; Poulícková *et al.* 2001; Tipping *et al.* 2002).

Macroinvertebrates and Diatoms as Biomonitors

It has been stated that macroinvertebrates are useful biomonitors of water quality due to their sensitivity (Dills & Rogers Jr 1974). The results of the current study partly support this statement with invertebrate species composition differing between the wetland groups. The lack of relationship between macroinvertebrate species richness and pH and the overlap of dominant species between the Group 1 and Group 2 wetlands both dispute the sensitive response of invertebrates to pH.

The major disadvantage of using macroinvertebrates as biomonitors may be that certain species do not respond to all environmental impacts. Invertebrate abundance and distribution can be influenced by factors other than water quality (Rosenberg & Resh 1993). For example physical factors such as channel width (Soininen & Könönen 2004), substratum (Sládecek *et al.* 1982), season (Lenat & Barbour 1994) and vegetation (Crowder & Cooper 1982) can all affect the species composition of invertebrates.

Aquatic diatom assemblages are known to reflect the water quality (Denys & van Straaten 1992). Diatoms are more sensitive to environmental variables than most other aquatic organisms (John 1998) and the strong relationship between diatom distribution and pH has been recognised for decades (Battarbee *et al.* 2001). The documented sensitivity of diatoms to changing pH is generally supported by the results of the study, with clear groupings of diatom assemblages in wetlands with differing pH. The differences in the diatom assemblages reiterate the well-defined ecological tolerances of diatoms and their sensitivity to changes in water chemistry such as decreasing pH (Charles 1985; Siver *et al.* 2004). The exhaustive autecological information available on diatom taxa should be an advantage for their use as ideal biomonitors.

Although dominant invertebrate and diatom taxa were identified for the Group 1 and the Group 2 wetlands, the results suggest that diatoms were more sensitive to acidity than were invertebrates. The overall community structure of invertebrates was influenced by pH but none of the dominant species were unique to wetlands displaying water quality similar to that of the Collie waterbodies. The composition of diatom assemblages was also affected by pH but, unlike the invertebrates, most dominant species of diatoms in the Group 1 waterbodies exhibited narrow tolerances to pH and could be used as biomonitors for acidity. The diatom assemblages in the Group 2 wetlands represent a shift in dominance. While most of the dominant species preferred waters below pH 7, Group 2 wetlands maintained a higher proportion of alkaliphilous species in comparison to the Group 1 wetlands. This was probably in response to the higher pH of Group 2 given that as pH moves closer to neutral, species with less defined tolerances can inhabit the waterbodies.

Further research will be helpful to fully examine the merits of macroinvertebrates and diatoms as biomonitors for acidic void wetlands. Future sampling regimes should be expanded to include the measurement of parameters that may influence macroinvertebrates such as peripheral vegetation and sediment characteristics. In terms of diatoms, an aspect that warrants further consideration is the lack of top down effect due to limited algal grazers, and how this might influence biomass and community composition. The incorporation of these variables would provide greater insight into the factors that may influence distribution of the two biomonitors.

Given that a healthy ecosystem comprises of abiotic characteristics and biotic communities (Loeb 1994) an integrated approach using macroinvertebrates and diatoms may be the most effective means of assessing acid waters such as the Collie wetlands. Currently the authors are developing a predictive model that will be applicable for the evaluation and management of similar mine-voids throughout Australia.

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References

Abel P D 1989 Water Pollution Biology. Ellis Horwood Ltd, Chichester.

Ashton P 1988 Exploration and geology. *In:* 100 Years of Collie Coal (ed C Stedman). Curtin Printing Services, Western Australia, 63–96.

Balla S A & Davis J A 1993 Managing Perth's Wetlands to Conserve the Aquatic Fauna. Wetlands of the Swan Coastal Plain Volume 5. Water Authority of Western Australia and Environmental Protection Authority.

Battarbee R W, Charles D F, Dixit S S & Renberg I 1999 Diatoms as indicators of surface water acidity. *In:* The Diatoms: Applications for the Environmental and Earth Sciences (eds E F Stoermer & J P Smol). Cambridge University Press, 85–127.

Battarbee R W, Flower R J, Juggins S, Patrick, S T & Stevenson A C 1997 The relationship between diatoms and surface water quality in the Høylandet area of Nord-Trøndelag, Norway. Hydrobiologia 348: 69–80.

Battarbee R W, Jones V J, Flower R J, Cameron N G, Bennion H, Carvalho L & Juggins, S 2001 Diatoms. *In:* Tracking Environmental Change Using Lake Sediments Volume 3: Terrestrial, Algal and Siliceous Indicators (eds J P Smol, H J B Birks & W M Last). Kluwer Academic Publishers, Dordrecht, 155–202.

Brugam R B & Lusk M 1986 Diatom evidence for neutralization in acid surface mine lakes. *In:* Diatoms and Lake Acidity (eds J P Smol, R W Battarbee, R B Davis & J Meriläinen). Dr W Junk Publishers, Dordrecht, 115–129.

Charles D F 1985 Relationships between surface sediment diatom assemblages and lakewater characteristics in Adirondack lakes. Ecology 66(3): 994–1011.

Cheal F, Davis J A & Growns J E 1993 Relationships between macroinvertebrates and environmental variables. *In:* Wetlands of the Swan Coastal Plain 6: Wetland Classification on the Basis of Water Quality and Invertebrate Community Data (eds J A Davis, R S Rosich, J S Bradley, J E Growns, L G Schmidt & F Cheal). Water Authority of Western Australia and Environmental Protection Authority, 171–190.

Chessman B C 1995 Rapid assessment of rivers using macroinvertebrates: a procedure based on habitat-specific sampling, family level identification and a biotic index. Australian Journal of Ecology 20: 122–129.

Chessman B, Growns I, Currey J & Plunkett-Cole N 1999 Predicting diatom communities at the genus level for the rapid biological assessment of rivers. Freshwater Biology 41: 317–331.

Courtney L A & Clements W H 1998 Effects of acidic pH on benthic macroinvertebrate communities in stream microcosms. Hydrobiologia 379: 135–145.

Crowder L B & Cooper W E 1982 Habitat structural complexity and the interaction between bluegills and their prey. Ecology 63(6): 1802–1813.

Davis J A & Christidis F 1997 A Guide to Wetland Invertebrates of Southwestern Australia. Western Australian Museum, Perth.

Davis J A & Rolls S W 1987 A Baseline Biological Monitoring Programme for the Urban Wetlands of the Swan Coastal Plain, Western Australia. Bulletin 265. Environmental Protection Authority, Perth, Western Australia.

DeNicola D M 1986 The representation of living diatom communities in deep-water sedimentary diatom assemblages in two Maine (USA) lakes. *In:* Diatoms and Lake Acidity (eds J P Smol, R W Battarbee, R B Davis & J Meriläinen). Dr W Junk Publishers, Dordrecht, 73–86.

DeNicola D M 2000 A review of diatoms found in highly acidic environments. Hydrobiologia 433: 111–122.

Denys L & van Straaten D 1992 A survey of the acid water diatom assemblages of two heathland relics in the Belgian Northern Campine (Groot & Klein Schietveld, Brasschaat) with an assessment of their conservational value. Diatom Research 7(1): 1–13.

Dills G & Rogers Jr D T 1974 Macroinvertebrate community structure as an indicator of acid mine pollution. Environmental Pollution 6: 239–262.

Eloranta P 1990 Ecology of some acidophilic and acidobiontic diatoms in Finnish forest lakes. *In:* Proceedings of the 10th International Diatom Symposium, Joensuu, Finland (ed H Simola). Koeltz Scientific Books, Koenigstein, 491–498.

Environmental Protection Authority 1992 Ewington Open Cut Mine, Collie. Bulletin 612. Environmental Protection Authority, Perth.

Foged N 1978 Diatoms in Eastern Australia. Bibliotheca Phycologica 41: 1–243.

Foged N 1979 Diatoms in New Zealand, the North Island. Bibliotheca Phycologica 47: 1–224.

Ford J 1986 The recent history of a naturally acidic lake (Cone Pond, NH). *In:* Diatoms and Lake Acidity (eds J P Smol, R W Battarbee, R B Davis & J Meriläinen). Dr W Junk Publishers, Dordrecht, 131–148.

Gambs G C & Walsh M A 1981 Surface mining of coal. *In:* Coal Handbook (ed R A Meyers). Marcel Dekker Inc., New York, 123–172.

Gasse F 1986 East African diatoms and water pH. *In:* Diatoms and Lake Acidity (eds J P Smol, R W Battarbee, R B Davis & J Meriläinen). Dr W Junk Publishers, Dordrecht, 149–168.

Growns J E, Davis J A, Cheal F & Bradley J S 1993 Classification of the wetlands using invertebrate data. *In:* Wetlands of the Swan Coastal Plain 6. Wetland Classification on the Basis of Water Quality and Invertebrate Community Data (eds J A Davis, R S Rosich, J S Bradley, J E Growns, L G Schmidt & F Cheal). Water Authority of Western Australia and Environmental Protection Authority, 88–156.

Hammer D A 1992 Creating Freshwater Wetlands. Lewis Publishers, Boca Raton.

Hawking J H & Smith F J 1997 Colour Guide to Invertebrates of Australian Inland Waters. Identification Guide No. 8. Co-operative Research Centre for Freshwater Ecology, Albury.

Haworth E, Atkinson K M & Carrick T R 1990 The preliminary assessment of diatom distribution in the waterbodies of Cumbria, North-West England. *In:* Proceedings of the 10th International Diatom Symposium, Joensuu, Finland (ed H Simola). Koeltz Scientific Books, Koenigstein, 459–470.

Hellawell J M 1986 Biological Indicators of Freshwater Pollution and Environmental Management. Elsevier Applied Science Publishers, Dordrecht.

Horwitz P 1995 A Preliminary Key to the Species of Decapoda (Crustacea: Malacostraca) found in Australian Inland Waters. Co-operative Research Centre for Freshwater Ecology, Albury.

Hustedt F 1937–1939 Systematische und ökologische Untersuchungen über den Diatomeen-flora von Java, Bali und Sumatra. Arch. Hydrobiologia (suppl.) 15/16.

Ingram B A, Hawking J H & Shiel R J 1997 Aquatic Life in Freshwater Ponds: a Guide to the Identification and Ecology of Life in Aquaculture Ponds and Farm Dams in South Eastern Australia. Identification Guide No. 9. Co-operative Research Centre for Freshwater Ecology, Albury.

John J 1983 The diatom flora of the Swan River Estuary Western Australia. Bibliotheca Phycologica 64: 1–359.

John J 1993 The use of diatoms in monitoring the development of created wetlands at a sandmining site in Western Australia. Hydrobiologia 269/270: 427–436.

John J 1997 Passive Mine Drainage Treatment and Vegetation Strategies. Report. Curtin University of Technology, Bentley.

John J 1998 Diatoms: Tools for Bioassessment of River Health. Land and Water Resources Research and Development Corporation, Canberra.

John J 2003 Bioassessment of Health of Aquatic Systems by the Use of Diatoms. *In:* Modern Trends in Applied Aquatic Ecology (eds RS Ambasht & NK Ambasht). Kluwer Academic/Plenum Publishers, New York, 1–20.

John J, Lowe E, Baxter L & Jones L 1999 A Photovoltaic Powered Passive Treatment System and Biomonitoring for Collie Mine Voids. Wetlands Research Group Report 1. Curtin University of Technology, Bentley.

Kalin M, Cao Y, Smith M & Olaveson M M 2001 Development of the phytoplankton community in a pit-lake in relation to water quality changes. Water Research 35(13): 3215–3225.

Kwandrans J 1993 Diatom communities of acidic mountain streams in Poland. Hydrobiologia 269/270: 335–342.

Lenat D R & Barbour M T 1994 Using benthic macroinvertebrate community structure for rapid, cost-effective water quality monitoring: rapid bioassessment. *In:* Biological Monitoring of Aquatic Systems (eds S L Loeb & A Spacie). Lewis Publishers, Boca Raton, 187–215.

Lepistö L 1988 Some centric diatoms as indicators of water quality in Finnish lakes. *In:* Proceedings of the 10th International Diatom Symposium, Joensuu, Finland (ed H Simola). Koeltz Scientific Books, Koenigstein, 499–504.

Loeb S L 1994 An ecological context for biological monitoring. *In:* Biological Monitoring of Aquatic Systems (eds S L Loeb & A Spacie). Lewis Publishers, Boca Raton, 3–7.

Lord J H 1979 Coal. *In:* Mining in Western Australia (ed R T Prider). University of Western Australia Press, Perth, 145–159.

Lydén A & Grahn O 1985 Phytoplankton species composition, biomass and production in Lake Gårdsjön – an acidified clearwater lake in SW Sweden. Ecological Bulletins 37: 195–202.

McNicol D K, Bendell B E & Mallory M L 1995 Evaluating macroinvertebrate responses to recovery from acidification in small lakes in Ontario, Canada. Water, Air and Soil Pollution 85: 451–456.

New T R 1995 An Introduction to Invertebrate Conservation Biology. Oxford University Press, Oxford.

Norris R H 1986 Mine waste pollution of the Molonglo River, New South Wales and the Australian Capital Territory: effectiveness of remedial works at Captains Flat mining area. Australian Journal of Marine and Freshwater Research 37: 147–157.

Patrick R & Reimer C W 1966 The Diatoms of the United States. Volume 1. The Academy of Natural Sciences of Philadelphia, Philadelphia.

Patrick R & Reimer C W 1975 The Diatoms of the United States. Volume 2, Part 1. The Academy of Natural Sciences of Philadelphia, Philadelphia.

Poulícková A, Duchoslav M, Hekera P, Hájková P & Novotny R 2001 Ecology of diatoms of sloping springs in the flyash area of the West Carpathians. *In:* Proceedings of the 16th International Diatom Symposium, Athens & Aegean Islands (ed A Economou-Amilli). University of Athens, Greece, 225–235.

Renberg I, Hellberg T & Nilsson M 1985 Effects of acidification on diatom communities as revealed by analyses of lake sediments. Ecological Bulletins 37: 219–223.

Rosenberg D M & Resh V H (1993). Introduction to freshwater biomonitoring and benthic macroinvertebrates. *In:* Freshwater Biomonitoring and Benthic Macroinvertebrates (eds D M Rosenberg & V H Resh). Chapman & Hall, New York, 1–9.

Sandin L, Dahl J & Johnson R K 2004 Assessing acid stress in Swedish boreal and alpine streams using benthic macroinvertebrates. Hydrobiologia 516: 129–148.

Schindler D W 1990 Experimental perturbations of whole lakes as tests of hypotheses concerning ecosystem structure and function. OIKOS 57: 25–41.

Schmidt L G & Rosich R S 1993 Physical and chemical changes and processes. *In:* Wetlands of the Swan Coastal Plain 6: Wetland Classification on the Basis of Water Quality and Invertebrate Community Data (eds J A Davis, R S Rosich, J S Bradley, J E Growns, L G Schmidt & F Cheal). Water Authority of Western Australia and Environmental Protection Authority, 29–80.

Siver P A, Ahrens T D, Hamilton P B, Stachura-Suchoples K & Kociolek J P 2004 The ecology of diatoms in ponds and lakes on the Cape Cod peninsula, Massachusetts, U.S.A. with special reference to pH. *In:* Proceedings of the 17th International Diatom Symposium, Ottawa, Canada (ed M Poulin). Biopress Ltd, Bristol, 335–357.

Sládecek V, Hawkes H A, Alabaster J S, Daubner I, Nöthlich I, de L G Solbe J F & Uhlmann D 1982 Biological examination. *In:* Examination of Water for Pollution Control, a Reference Handbook, Vol. 3: Biological, Bacteriological and Virological Examination (ed M J Suess). Pergamon Press, Oxford, 1–272.

Soininen J & Könönen K 2004 Comparative study of monitoring South-Finnish rivers and streams using macroinvertebrate and benthic diatom community structure. Aquatic Ecology 38: 63–75.

Sommer B & Horwitz P 2001 Water quality and macroinvertebrate response to acidification following intensified summer droughts in a Western Australian wetland. Marine and Freshwater Research 52: 1015–1021.

Stokes P M & Yung Y K 1986 Phytoplankton in selected LaCloche (Ontario) lakes, pH 4.2–7.0, with special reference to algae as indicators of chemical characteristics. *In:* Diatoms and Lake Acidity (eds J P Smol, R W Battarbee, R B Davis & J Meriläinen). Dr W Junk Publishers, Dordrecht, 57–72.

Stronach B 1988 Open-cut mining. *In:* 100 Years of Collie Coal (ed C Stedman). Curtin Printing Services, Western Australia, 167–200.

Tipping E, Bass J A B, Hardie D, Haworth E Y, Hurley M A & Wills G 2002 Biological responses to the reversal of acidification in surface waters of the English Lake District. Environmental Pollution 116: 137–146.

Watanabe T & Asai K 2004 *Nitzschia paleaeformis* and *Nitzschia amplectens* occurring in strongly acid waters of pH range from 1.0 to 3.9 in Japan. Diatom 20: 153–158.

Williams W D 1980 Australian Freshwater Life. MacMillan Education Australia Pty. Ltd., South Yarra, Victoria.

Williams W D & Barnard J L 1988 The taxonomy of Crangonyctoid Amphipoda (Crustacea) from Australian fresh waters: foundation studies. Records of the Australian Museum Supplement 10: 1–180.