

Subsidence from an Underground Coal Mine and Mine Wastewater Discharge Causing Water Pollution and Degradation of Aquatic Ecosystems

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Abstract This study examined a single underground coal mine and investigated two aspects of its operation: the disposal of the mine waste through a discharge to a nearby river and the impact of subsidence from an underground longwall to a small waterway above. Water quality of the two waterways was monitored over a 2-year period with a monthly investigation over a 6-month period, which included collection of stream macroinvertebrates. Both mine activities modified surface water geochemistry and macroinvertebrate communities. Mean electrical conductivity (EC) increased in surface waters below the mine discharge, rising 4.8 times from (186 $\mu\text{S}/\text{cm}$) upstream to 1078 $\mu\text{S}/\text{cm}$ below the waste inflow. Mean EC increased in a small stream that was disturbed by subsidence from longwall mining, rising 3.8 times from (247 $\mu\text{S}/\text{cm}$) upstream to 1195 $\mu\text{S}/\text{cm}$ below. The mineral constituents of the increased salinities were different. The coal mine wastewater discharge was enriched with sodium and bicarbonate ions compared to sodium and chloride ions in the subsidence affected creek. Both the waste discharge and the subsidence caused increases in the concentrations of zinc by about four times and nickel by 20 to 30 times the background levels. The subsidence reduced dissolved oxygen to ecologically stressful levels and increased iron and manganese concentrations by about 20 times the background levels. Two of the key changes in stream

ecosystems were a reduction in the proportion of mayfly larvae downstream of the mine waste discharge and mosquito larvae dominating (60–70 % of total abundance) the invertebrate community in the subsidence affected creek.

Keywords Mining · Macroinvertebrates · Freshwater · Heavy metal · Zinc · Nickel · EC · pH · Pollution · Longwall · Culicidae · Water quality · Abundance · EPT richness · Ionic composition

1 Introduction

Coal mining can pollute ground and surface waters (Johnson 2003; Younger 2004). Water pollution from disposal of coal mine wastewater to local waterways often causes contamination of water which contributes to degradation of stream ecosystems (Jarvis and Younger 1997; Pond et al. 2008; Tiwary 2000). One well-known form of water pollution often associated with coal mining is *acid mine drainage* (AMD). Coal wastewater can be susceptible to AMD, depending on its sulphur content and disturbance linked to mining activity can trigger oxidation of the sulphur to generate sulphuric acid (Johnson 2003). An effect of the AMD sulphuric acid production is a reduction of pH and dissolution and mobilisation of metals and minerals from the surrounding geology (Johnson 2003). Water pollution attributed to AMD and coal mine wastes discharged to surface waters is a worldwide environmental coal mining problem and often includes changes

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to river and stream pH (Verb and Vis 2000; Brake et al. 2001; Banks et al. 1997), elevated electrical conductivity (EC) (García-Criado et al. 1999; Pond et al. 2008), modified stream ionic composition (Wright 2012) and elevated heavy metals (Pond et al. 2008; Brake et al. 2001; Wright and Burgin 2009a, b; Johnson 2003).

A less studied form of water pollution, associated with coal mining, is caused by subsidence from underground longwall coal mining. The process of long wall mining generates surface subsidence due to movement and fracturing of geological strata following the removal of the coal seam. This can cause changes to surface and groundwater hydrology (Booth and Bertsch 1999; Jankowski 2007). Some of the water chemistry changes to ground and surface waters resulting from coal mining subsidence includes depletion of dissolved oxygen, alteration of pH, increased EC and elevated concentrations of metals such as iron, manganese, aluminium, zinc and nickel (Pigati and López 1999; Jankowski 2007). An abandoned underground coal mine in Ohio (USA) suffered subsidence following the collapse of internal mine supports, and this resulted in a surface stream being 'captured' through inflow to the mine workings. This resulted in increased flows of contaminated mine drainage emanating from the mine (Pigati and López 1999). The use of longwall coal mining, in particular, and associated subsidence has triggered environmental issues such as the loss of stream flow and the draining of wetlands of conservation significance (Krogh 2007). Degradation of waterways by longwall coal mining in Sydney's southern drinking water catchments, in the vicinity of rivers and streams, has resulted in widespread cracking of bedrock sections of stream channels and swamps (Krogh 2007; Jankowski 2007). In places, the lost streamflow in fractured sections has flowed into near-surface aquifers and has returned to the surface channel further downstream with deoxygenated water, elevated metal levels causing damage to fish and aquatic ecosystems (Jankowski 2007).

Elevated heavy metal concentrations in coal mine wastewater discharges have been reported from many studies across the world. Banks et al. (1997) examined water quality below a selection of abandoned and active British and Scandinavian mines and revealed a very large variation in metal concentrations when compared to background levels, particularly for iron, aluminium, manganese, zinc and copper. Brake et al. (2001) found highly acidic (pH 2.2 to 4.6) coal mine drainage (from

coal mines and mine spoil) in Indiana, USA. This was associated with very high levels of zinc, lead, chromium and nickel at levels hundreds of times higher than local uncontaminated reference streams. Pond et al. (2008) reported only modest increases in zinc, nickel and lead in streams affected by coal mine waste water discharge when compared to streams not exposed to mine waste water discharges. Increased metals in Waratah Rivulet, due to longwall subsidence, included manganese, iron, barium and strontium (Jankowski 2007). The subsidence affected Ohio coal mine released increased flows of contaminated mine drainage including (in order of loadings): iron (54 tonnes), sulfate (415 tonnes), aluminium (1 tonne), zinc (52 kg), nickel (22 kg) and cobalt (12 kg) (Pigati and López 1999).

There have been relatively few Australian studies that have investigated water pollution from coal mines and evaluated any related degradation of freshwater ecosystems. Battaglia et al. (2005) detected low pH (5.1) in a stream receiving mine drainage and concluded that the elevated zinc (*c.* 500 µg/L) and nickel (*c.* 400 µg/L) were both toxic and contributed to degradation of stream macroinvertebrate assemblages. Wright and Burgin (2009a, 2009b) reported elevated zinc levels from drainage flowing from an abandoned coal mine into a high conservation value river in the Sydney basin. The mine drainage lifted background concentrations of zinc from 10 to 440 µg/L and impaired downstream stream ecosystems with reductions in macroinvertebrate taxonomic richness and abundance (Wright and Burgin 2009a, 2009b). A similar study by Belmer et al. (2014) reported that a coal mine wastewater discharged increased EC, pH, nickel and zinc levels and reduced macroinvertebrate taxonomic richness and abundance downstream of the mine discharge.

In this current study, we use quantitative macroinvertebrate surveys and temporally replicated water quality sampling to detect and measure any adverse ecological and water quality effects from the operation of a single coal mine. In particular, we compared the impact of two aspects of the mine operation: the disposal of the mine's wastewater to the Bargo River and subsidence in Redbank Creek linked to longwall mining. In doing so, two questions have been posed:

1. How do stream macroinvertebrates respond to coal mine waste water discharges compared with longwall subsidence of streams?

2. What are the key water quality changes (particularly ionic composition and metals) to the stream downstream of the mine discharge compared to the subsidence affected stream?

2 Methods

2.1 Study Area

Samples were collected from two waterways (Redbank Creek and Bargo River) in the Hawkesbury-Nepean catchment in the Tahmoor, Thirlmere and Picton area of New South Wales in south-eastern Australia ($34^{\circ} 11' S$, $150^{\circ} 35' E$; Fig. 1). The study area comprised areas of natural vegetation, rural holdings (mostly livestock grazing) and urban townships.

The Tahmoor Colliery has operated since 1979 (Tahmoor Coal 2014). It is an underground coal mine operation that was given approval to begin longwall mining in 1986 (Tahmoor Coal 2014). Wastewater from the mine is generated by the entire mining operation through underground coal mining operations and coal washing and stockpiling at the mine surface. The discharge of waste water from the mine to the Bargo River is regulated by the NSW Environment Protection Authority under the *Protection of the Environment Operations* (1997) Act (POEO Act 1997).

Five sampling sites were selected in the study (Fig. 1). Three sites were chosen on Redbank Creek; the waterway subject to longwall mining below it. Site RB1 was within the township of Thirlmere, about 1 km upstream of the longwall mine and subsidence area (Fig. 1). The second site RB2 was on Redbank Creek above 'Longwall 26'. This section of creek included several reaches with a sandstone bedrock channel, and the occurrence of recent fracturing from subsidence was obvious. RB2 was established in this vicinity of stream bedrock fracturing where some groundwater flow remerged in the creek channel as surface water. A third sampling site (RB3) was located above 'Longwall 27', about 100 m downstream, of RB2. This site was chosen due to the presence of a large stream pool, and this section of creek appeared to have a more reliable flow than sections further upstream.

Two sampling sites were located on the Bargo River which was the waterway used to dispose of mine wastewater from the Tahmoor coal mine (Fig. 1). One site

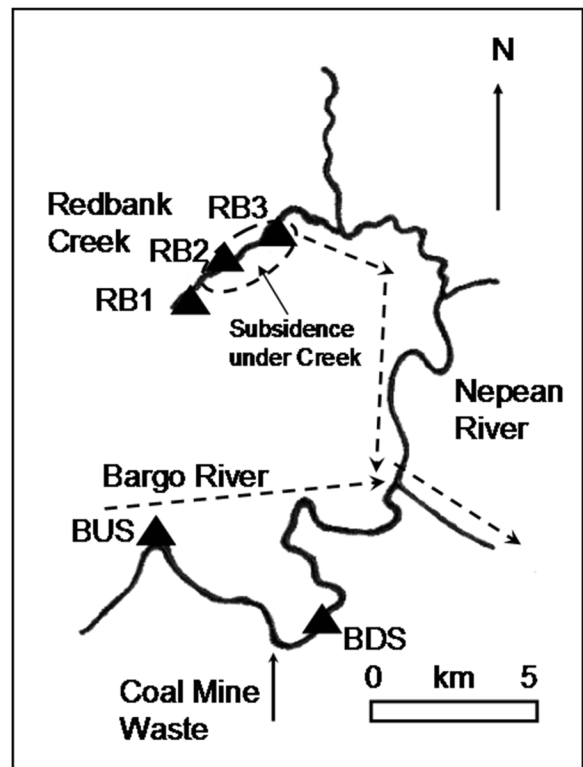


Fig. 1 Map of sampling sites, represented by black triangles, on Redbank Ck (RB1 upstream of subsidence) and the two sites (RB2 and RB3) located in the section of creek affected by subsidence. Also shown are the two sites on Bargo River (BUS upstream and BDS downstream of mine waste discharge). Hyphenated arrows represent direction of waterway flow

Bargo upstream (BUS) was about 1 km above the coal mine waste discharge, and the second site Bargo downstream (BDS) was approximately 2 km below the inflow of coal mine wastewater to the Bargo River. No major inflows, except the mine wastewater, enter the Bargo River between these two sites.

2.2 Macroinvertebrate Sampling

The collection of stream and river macroinvertebrates to assess the ecological effects of water pollution is widely used due to their ease of sampling, relatively long life cycles and sensitivity to disturbance and water pollution (Hellowell 1986). Evaluation of freshwater macroinvertebrates to measure ecological distress has been applied to a wide variety of human-induced pollution sources (Hynes 1960; Rosenberg and Resh 1993) including coal mine drainage (Jarvis and Younger 1997; Battaglia et al. 2005; Pond et al. 2008).

Macroinvertebrates were collected from all sites in May 2014, and then a second round of sampling was conducted in August 2014 (Bargo River) and September 2014 (Redbank Creek).

When collecting macroinvertebrates, five quantitative benthic samples were collected from pool edge habitat (Resh and Jackson 1993; Wright et al. 1995) in Redbank Creek as it was the only widely available habitat present at each sampling location. Bedrock riffle zone was the habitat sampled in Bargo River for the same reasoning that it was the most widely available habitat. The location of each replicate was randomly selected within a 15-m stream reach. A comparable macroinvertebrate sampling technique was used in a similar earlier study (of coal mine and sewage point sources) by the senior author of the Grose River (Wright and Burgin 2009a) and in a coal mine study of the Wollangambe River (Belmer et al. 2014).

Samples were collected using a kick sampling technique. A net with a frame of 30×30 cm and 250 µm mesh was used (Rosenberg and Resh 1993; Wright and Burgin 2009a). Sampling was achieved by disturbing the stream bottom for a period of 30 s over a 900 cm² area, immediately upstream of the net. The net contents, including stream detritus and macroinvertebrates, were immediately placed into a sealed and labelled storage container and preserved in 70 % ethanol.

In the laboratory, the remaining material was then sorted under a dissecting microscope (×10 to ×60) to extract the macroinvertebrates from stream detritus (e.g. leaves, sticks, rocks and gravel). All insect groups were identified to family as these data have been demonstrated to provide adequate taxonomic resolution for impact assessment (Wright et al. 1995). Two non-insect groups (Oligochaeta, Hydracarina) were not identified to the family level due to identification difficulties.

2.3 Biotic Indices

There have been many biotic indices developed to help interpret stream macroinvertebrate results to detect and measure the ecological condition of waterways. Two of the most commonly and simply calculated indices are taxa richness and total abundance [(see Resh and Jackson 1993)]. Abundance is often ignored due to the proliferation of qualitative rapid assessment methodologies [(e.g. Resh and Jackson 1993; Chessman 1995)], but two previous studies of coal mines found that abundance of macroinvertebrates (at both the community and

family level) provided very important information that help interpret the ecological impact of the mine waste (Wright and Burgin 2009a; Belmer et al. 2014). The EPT (Ephemeroptera, Plecoptera and Trichoptera) index is a commonly used biotic index based on the relative abundance or taxonomic richness of three common macroinvertebrate orders that have demonstrated sensitivity to disturbance and degraded water quality (Lenat and Penrose 1996). EPT taxa richness was one of the most sensitive biotic indices for evaluating ecological damage from coal mine pollution emitted from a non-operational coal mine in the nearby Grose valley (Wright and Burgin 2009a).

2.4 Water Quality Sampling

Water sampling was conducted at three sites on Redbank Creek on 12 occasions at all Redbank sites over a 2 ½ year period (June 2012 to December 2014). Sampling of Redbank Creek was first conducted at RB1 and RB2 in June 2012 when longwall mining (Longwall 26) first passed under Redbank Creek, in the vicinity of RB2. On the next sampling occasion (10 months later), stream flow was absent from many sections of Redbank Creek, but water was available, albeit at a very low rate of flow, at all three sites (RB1, RB2 and RB3). The third sampling occasion was July 2013 then December 2013 and April 2013. A more intense period of water sampling was conducted at all sites from April to October 2014 at monthly intervals, with a final sample collected in December 2014. Water sampling was conducted at two sites on the Bargo River on 11 occasions over a 1 ½ year period (July 2013 to December 2014).

At each site, on each sampling occasion, physiochemical water quality attributes of pH, EC, dissolved oxygen (DO) and water temperature were measured in-situ using a TPS AQUA-Cond-pH meter (for pH and EC) and a YSI ProODO meter (dissolved oxygen and water temperature) and a HACH 2100P turbidimeter for turbidity. The calibration of each meter was checked on each day and adjusted if necessary. Five replicated measurements of each of the above water quality attributes were recorded from each site on each sampling occasion. Grab samples were collected in decontaminated sample containers provided by a commercial testing laboratory. Samples were chilled and delivered to the laboratory for analysis. All grab samples were analysed using standard methods (APHA 1998) by a National Associations of Testing Authorities (NATA) accredited

laboratory for major anions, major cations and total metals (zinc, nickel, aluminium, manganese and iron).

2.5 Data Analysis

Multivariate analyses of macroinvertebrate community data were conducted to investigate the ecological response, if any, of stream macroinvertebrates at the different sampling sites (Wright et al. 1995). Non-metric multidimensional scaling (nMDS) and analysis of similarity (ANOSIM) are a non-parametric technique that has proven very effective in assessing ecological responses to human perturbations, including assessing marine and freshwater pollution (Clarke 1993; Warwick 1993). nMDS was performed on the similarity matrix, computed with square-root transformed macroinvertebrate taxon abundance data, using the Bray-Curtis dissimilarity measure (Clarke 1993; Warwick 1993). The taxon abundance data was square root transformed to reduce the influence of rare or common taxa in the analysis of community structural differences at multiple sampling sites (Wright et al. 1995). Two-dimensional ordination plots represented the dissimilarity among samples. The data analysis was conducted separately on the samples collected from the Bargo River and Redbank Creek. This was due to different habitat conditions being present at each waterway (pool edge for Redbank Creek and riffle for Bargo River). A total of 10 replicates were grouped for each upstream site (Bargo upstream and RB1) as reference samples to test for differences by one-way analysis of similarity (ANOSIM: Clarke 1993) between (upstream) reference sites and sites downstream of the waste discharges. For the Bargo River, one site downstream was sampled twice (10 replicates), and for Redbank Creek, two downstream sites were sampled twice (total of 20 replicates). These multivariate analyses were achieved using the software package PRIMER version 5 (Clarke 1993).

For univariate data analysis, each parameter (water chemistry and biotic indices) was evaluated for heterogeneity of variance using a Levene's test. Since most were abnormally distributed, all were analysed using a non-parametric technique. The Mann-Whitney *U* test was used for Bargo River (upstream versus downstream), and the Kruskal-Wallis (non-parametric) test was used to compare differences across the three sampling sites for Redbank Creek. All univariate statistical analyses were performed using IBM SPSS Statistics version 22.

3 Results

3.1 Water Chemistry

Water chemistry results show that each of the coal mine activities (waste disposal and subsidence) caused different changes to the chemical properties of each waterway (Tables 1 and 2). EC increased in both waterways (Table 1). The lowest EC level in the study was recorded in the Bargo River above the mine (mean 186.2 $\mu\text{S}/\text{cm}$) which increased to a mean of 1078 $\mu\text{S}/\text{cm}$ below the mine (Table 1). Mean EC levels in Redbank Creek, ranged from 247.3 $\mu\text{S}/\text{cm}$, upstream of the subsidence, rising to 406.2 and 1195.4 $\mu\text{S}/\text{cm}$ at the two sites affected by subsidence (Table 2).

Mean water pH at the two reference (upstream) sites was very similar (6.54 Bargo River and 6.79 Redbank Creek). The pH trends diverged with the most downstream sampling site (RB3) affected by subsidence had a mean pH of 5.97 and the mean pH below the waste discharge in Bargo River was 8.19 (Tables 1 and 2).

Dissolved oxygen levels were much higher in the Bargo River than Redbank Creek (Fig. 1). DO was highest in the Bargo River above the coal mine (mean 96.5 % saturation) dropping to a mean of 89.5 % below the mine waste (Table 1). In comparison, the mean DO levels in Redbank Creek were 51.1 % above the subsidence, falling to 39.2 % at RB2 and RB3 (mean 21.9 %) in the subsidence zone (Table 2).

Bicarbonate concentrations also showed contrasting trends in the two waterways (Fig. 1). The largest change was recorded in the Bargo River where the mean bicarbonate concentration increased by 45 times (Table 1) rising from 9.5 mg/L, above the coal mine, to 439.2 mg/L below the mine waste inflow (Table 1). In comparison, the mean bicarbonate concentrations fell in Redbank Creek from 46.4 mg/L above the subsidence to 34.1 and 9.2 mg/L at the two sites in the subsidence zone (Table 2). Only at one site was bicarbonate the dominant anion (Bargo River below the mine) at all other sites it was sub-dominant (Tables 1 and 2).

Sodium was the dominant cation at all sampling sites in this study (Tables 1 and 2). The sodium concentration varied highly significantly at sampling sites in the Bargo River (Table 1). The largest change was recorded in the Bargo River where the mean sodium concentration increased by more than 10 times (Table 1). It had a mean concentration of 23.3 mg/L, above the coal mine, and 252 mg/L below the mine (Table 1). The mean sodium

Table 1 Macroinvertebrate biotic index and water quality variables from the Bargo River, collected upstream (BUS) and downstream (BDS) of the coal mine waste discharge. Results for Mann-Whitney U-test *p* values are provided (significance differences

appear in bold), along with the range, mean and median for each variable to test for variation upstream compared to downstream of the mine discharge. (bd=below detection)

Source of variation	Mann-Whitney U test <i>p</i> value	Upstream (BUS)		Downstream (BDS)	
		Range	Mean (Median)	Range	Mean (Median)
Biotic indices					
EPT %	0.190	6.86–95.4	55.0 (73.4)	6.06–77.7	37.2 (28.8)
Ephemeroptera %	0.159	1.2–41.9	15.0 (9.8)	0–15.7	4.2 (1.8)
Culicidae %	0.739	0–3	0.3 (0)	0	0 (0)
Richness	0.859	5–14	8.8 (8.5)	4–10	7 (7)
Abundance	0.961	68–933	231.6 (168.5)	11–721	196.8 (154.5)
Water quality					
pH (pH units)	<0.0001	5.2–6.99	6.54 (6.7)	6.33–8.8	8.19 (8.29)
Salinity (µS/cm)	<0.0001	140.1–235.3	186.2 (201.8)	345–1741	1078 (1106)
Dissolved oxygen (% saturation)	0.334	91.7–105.4	96.5 (95.6)	81.5–98.8	89.5 (90.4)
Water Temp. ° C.	<0.0001	9.5–21.5	14.48 (13)	8.3–22	14.96 (14.8)
Turbidity NTU	0.564	0.8–13.7	5.9 (5.19)	1.41–40.4	12.2 (5.96)
Carbonate (mg/L)	–	bd	–	bd–109	36.8 (22)
Bicarbonate (mg/L)	0.315	5–13	9.5 (10)	120–720	439.2 (460)
Chloride (mg/L)	<0.0001	30–57	44.8 (44.5)	32–62	48.6 (52.5)
Sulfate (mg/L)	0.003	4–6	4.6 (4.5)	5–16	9.75 (9)
Calcium (mg/L)	<0.0001	2–3.2	2.69 (2.6)	6–16	11.27 (11.5)
Potassium (mg/L)	<0.0001	1.8–2.8	2.22 (2.1)	5.7–20	13.96 (15.5)
Magnesium (mg/L)	<0.0001	3–6	4.36 (4)	5.3–13	9.28 (10)
Sodium (mg/L)	<0.0001	16–31	23.3 (26)	72–430	252 (257)
Nickel (µg/L)	0.918	BD–2	1.65 (2)	7–82	35.6 (33)
Zinc (µg/L)	<0.0001	2–11	7.9 (9)	12–72	38.6 (34.5)
Manganese (µg/L)	0.101	74 - 168	122.9 (125)	12–45	28.2 (28.5)
Aluminium (µg/L)	0.434	5–220	71 (55)	50–560	163.7 (120)
Iron (µg/L)	0.918	120–2700	1574 (1700)	81–1000	471.4 (410)

concentrations also increased in Redbank Creek from 42.6 mg/L above the subsidence to 44.2 and 176.2 mg/L at the two subsidence affected sites (Table 2).

Chloride was the dominant anion at all sampling sites in this study, except for Bargo River below the mine waste discharge (Tables 1 and 2). The chloride concentrations were similar at both sites in the Bargo River, ranging from 30 to 62 mg/L. Chloride concentrations were higher in Redbank Creek with mean concentrations of 75.8 and 72.2 mg/L at the upper two sites, and the highest concentration (mean 327.8 mg/L) was recorded at the lowest site in the subsidence zone (Table 2).

Zinc concentrations in the Bargo River increased from a mean of 7.9 µg/L above the mine to a mean 38.6 µg/L below the mine (Table 1). The mean zinc concentrations in Redbank Creek increased from 15.4 and 15.6 µg/L at the two upper sites and was highest (mean 86.3 µg/L) at the most subsidence affected site (Table 2). Nickel concentrations in the Bargo River increased from a mean of 1.65 µg/L above the mine to 35.6 µg/L below the mine (Table 1). The mean nickel concentrations in Redbank Creek progressively increased from the site above the subsidence (mean 0.9 µg/L) to the two sites in the subsidence zone (RB2 mean 4.5 µg/L) and were highest at the lowest site (mean 30.5 µg/L) at RB3 (Table 2).

Table 2 Macroinvertebrate and water quality variables from Redbank Creek, collected at three sites (RB1, RB2 and RB3) with the RB2 and RB3 subject to subsidence. Results for Kruskal-Wallis *p* values are provided (significance differences appear in

bold), along with the range, mean and median for each variable testing for spatial differences within Redbank Creek. (bd=below detection).

Source of variation	Kruskal-Wallis <i>p</i> values	Redbank Creek					
		RB1		RB2		RB3	
		Range	Mean (med.)	Range	Mean (med.)	Range	Mean (med.)
Biotic indices							
Culicidae %	0.001	0–1.9	0.64 (0.27)	47.8–96.7	67.0 (63.4)	15–93.2	60.4 (75.9)
Ephemeroptera %	0.006	0–0.79	0.14 (0)	0–37.9	9.7 (5.0)	0–15.8	3.7 (2.9)
EPT %	0.001	0–0.8	0.19(0)	0.4–37.3	12.9 (8.7)	0–15.8	5.0 (4.4)
Richness	0.015	7–16	10.5 (9.5)	4–14	7.6 (7.0)	4–12	7.3 (6.5)
Abundance	0.670	39–364	151.8 (111.5)	33–566	154 (88.0)	12–132	57.4 (50.0)
Water quality							
pH (pH units)	<0.0001	5.9–7.9	6.79 (6.83)	5.19–7.85	6.63 (6.67)	5.07–7.39	5.97 (5.87)
Salinity (µS/cm)	<0.0001	163–463	247.3 (237.4)	140–1013	406.2 (308.5)	155.8–2381	1195.4 (1439)
Dissolved oxygen (% saturation)	<0.0001	7.1–116.5	51.1(45.9)	5.8–75	39.2 (41.2)	2 – 111.1	21.9 (10.6)
Turbidity (NTU)	<0.0001	6.22–36.5	13.0 (9.59)	1.06–40.9	14.4 (13)	8.07 -35.5	22.3 (22.2)
Bicarbonate (mg/L)	0.005	bd–104	46.4 (47.5)	23–45	34.1 (31)	bd–25	9.2 (6.2)
Chloride (mg/L)	0.135	17–417	75.8 (46.5)	19–200	72.2 (42.9)	27–500	327.8 (385)
Sulfate (mg/L)	0.025	3 - 16	10.4 (11.5)	bd–23	11.1 (9)	4–9	5.7 (5)
Calcium (mg/L)	0.333	9–28	14.7 (14)	5.6–26	13.4 (12)	6.5–22	14.6 (15.5)
Potassium (mg/L)	0.148	1.5–14	4.84 (4)	4.9–29	10.8 (7.25)	4.4–7.1	5.53 (5.1)
Magnesium (mg/L)	0.112	2.8–40	8.5 (6)	3.7–29	12.4 (8.5)	4.3–57	37 (42.5)
Sodium (mg/L)	0.177	16–176	42.6 (31.5)	14–84	44.2 (37.5)	17–280	176.2 (205)
Nickel	<0.0001	bd–3	0.9 (1)	bd–31	4.5 (1)	2–110	30.5 (18.5)
Zinc	0.071	bd–36	15.4 (14)	bd–77	15.6 (9.5)	13–300	86.3 (52)
Aluminium	0.198	40–11,000	1013 (200)	40–540	191.4 (170)	5–440	99.2 (30)
Manganese	0.146	5–1800	251 (66)	2–2370	423.6 (18)	28–17,000	4954 (3100)
Iron	0.146	230–14,300	2816 (1080)	120–14,800	2013 (700)	380–82,000	20,163 (9700)

Iron concentrations changed in both waterways differently (Tables 1 and 2). The mean iron concentration in Redbank Creek increased from RB1 (2816 µg/L) and RB2 (2013 µg/L) and was highest (20163 µg/L) at RB3. The trend was opposite in the Bargo River with iron concentration dropping from a mean of 1574 µg/L above the mine to 471.4 µg/L below the mine. Similarly, manganese concentrations followed the same pattern at the two waterways, increasing in the presence of subsidence from RB1 above the subsidence (mean 251 µg/L) to subsidence impacted RB3 (mean 4954 µg/L) (Table 2).

3.2 Macroinvertebrates

A total of 7962 aquatic invertebrates were collected and identified from the two waterways and five sampling sites in this study. The collection comprised of 52 different taxonomic groups, mostly families. The majority (85.8 %) of the invertebrates were insects. Non-biting midge larvae (family Chironomidae) were the most commonly collected taxonomic group collected and accounted for just below 50 % of all invertebrates. The second most commonly collected group were mosquito larvae (family Culicidae), which included 20.4 % of

invertebrates collected. Approximately 10 % of all invertebrates collected were from the sensitive EPT insect orders.

Multivariate nMDS analysis revealed that macroinvertebrate assemblages were modified at the sampling sites subject to each of the two mine activities (waste disposal in Bargo River and subsidence in Redbank Creek)(Figs. 2 and 3). Community structure varied highly significantly according to sampling site (Global $R=0.618$, $p<0.001$) in Redbank Creek. Pairwise comparisons showed that the invertebrate assemblages at each of the sites in the subsidence impact zone (RB2 and RB3) were highly dissimilar ($R=0.872$, $p=0.1\%$ and $R=0.893$, $p=0.1\%$) to the upstream (reference) site. Macroinvertebrate samples from the two sites in the subsidence zone were not ecologically dissimilar ($R=0.019$, $p=34.6\%$) from each other. In comparison, the ecological differences were more subtle ($R=0.166$, $p=2.8\%$) upstream of the mine versus downstream in the Bargo River.

The two-dimensional nMDS ordination of macroinvertebrate samples from Redbank Creek showed the formation of two clusters of samples (Fig. 2). The cluster, in the left of the nMDS, is the group of Redbank Creek samples from RB1, upstream of the long-wall mining activity (Fig. 2). The second cluster comprises samples from the two sampling sites (RB2 and RB3) within the subsidence zone (Fig. 2).

The two-dimensional nMDS ordination of macroinvertebrate samples from Bargo River showed the formation of two clusters of samples (Fig. 3), with some overlapping of the two clusters. The cluster, to the left of the nMDS, is the group of Bargo River samples from

upstream of the coal mine waste discharge with the downstream samples to the right; Fig. 3).

At Redbank Creek the three biotic indices (Taxonomic richness, % EPT, % Ephemeroptera) varied significantly according to site location (Table 2). The mean taxonomic richness was highest at the upstream site RB1 (mean=10.5) and was lower at the two subsidence affected sites (mean 7.6 and 7.3; RB2 and RB3) (Table 2). The upstream (urban site; RB1) had the lowest proportion of EPT and Ephemeroptera (mean % EPT 0.19 % and mean Ephemeroptera 0.14 %), with highest mean levels at RB2 (mean EPT 12.9 % and Ephemeroptera 9.7 %) and lower proportions at RB3 (mean EPT 5.0 %, mean Ephemeroptera 3.7 %). The proportion of EPT animals was higher at both Bargo River sites than at any in Redbank Creek (Tables 1 and 2). No biotic index for Bargo River samples (upstream compared to downstream) varied significantly (Table 1).

Culicidae larvae were very uncommon in Redbank Creek above the subsidence (<1 %) but were the single most numerous invertebrate at the sites affected by subsidence where they accounted for a mean of 60 to 67 % of all invertebrates collected (Table 2). In comparison, Culicidae larvae were very rare in the Bargo River and were never collected below the mine (Table 1).

4 Discussion

This study found that a single coal mining operation contaminated local surface waters through the two aspects of its operations: the release of waste to a river and

Fig. 2 nMDS ordination of all Redbank Ck macroinvertebrate samples. Upstream samples (RB1) are represented by *squares*, and sites in the upper subsidence zone (RB2) are represented by *grey triangles* and further downstream in the subsidence zone (RB3) are represented by *unshaded triangles*

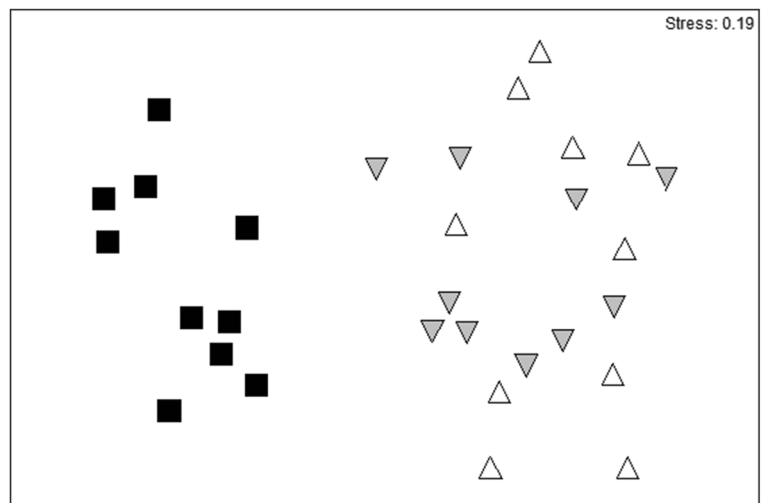
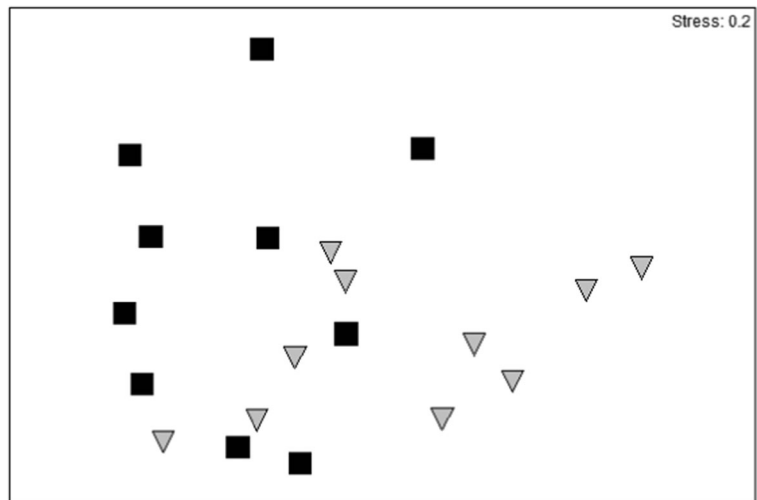


Fig. 3 nMDS ordination of all Bargo River macroinvertebrate samples. Samples are represented by *black squares* for samples from upstream (BUS) and *grey triangles* (BDS) from downstream of mine



subsidence from underground mining. Each activity was associated with different changes to physical, chemical and ecological properties of the two waterways.

Macroinvertebrate results showed that aquatic ecosystems were both adversely affected by each of the two mine operations. Our results showed that coal mine subsidence modified stream macroinvertebrate communities to a much greater degree than did the coalmine waste discharge; however, water quality and macroinvertebrates at the Redbank upstream site (RB1) were itself degraded in a pattern indicative of the *urban stream syndrome* in the Sydney basin (Tippler et al. 2012). The ANOSIM and nMDS results revealed highly dissimilar macroinvertebrate assemblages at the upstream reference site in Redbank Creek (RB1) compared to the lower two subsidence affected sites (RB2 and RB3). Three biotic indices varied significantly in Redbank Creek (taxonomic richness, % EPT and % Ephemeroptera). The taxonomic richness of macroinvertebrates was lowest at RB3, which also had the most impaired water quality. The same site also recorded a very low proportion of sensitive EPT invertebrates.

An unexpected change in the macroinvertebrate community was the very high proportion, ranging from 15 to 97 %, of Culicidae (mosquito) larvae dominating the macroinvertebrate communities at the subsidence-affected sites on Redbank Creek. The species of mosquito was not determined, but any changes to waterway flow or water quality that favours mosquitos could create a public health hazard. Other studies have shown that mosquitos have been detected in highly polluted waters, including the disease vector *Aedes aegypti* in

disused gold mine shafts in the Charters area of Queensland, Australia (Russell et al. 1996) and in heavy metal contaminated urban environments in Kenya (Mireji et al. 2008).

The macroinvertebrate community composition in the Bargo River changed below the mine waste discharge compared to that above the mine. Multidimensional scaling and ANOSIM analysis also confirmed that the community composition below the mine waste was different to samples collected upstream. Three other biotic indices (% EPT; family richness and abundance) did not vary significantly (upstream vs downstream) of the waste inflow. These results indicated that the level of ecological impairment resulting from the mine waste disposal into Bargo River was relatively mild compared to the impact of subsidence on Redbank Creek ecosystems and also compared to other mine waste studies (e.g. Pond et al. 2008). In the current study, the proportion of Ephemeroptera in Bargo River accounted for a mean of 15 % of the macroinvertebrate community upstream compared to 4.2 % downstream of the waste discharge. A comparatively larger impact on stream Ephemeroptera was reported in a study of coal mine impacts on West Virginia (USA) streams (Pond et al. 2008) which reported that the proportion of Ephemeroptera in unmined site averaged 45.6 % compared to 7.4 % at mined sites. The same study also reported significant reductions of family richness, % EPT and many other biotic indices at mined compared to unmined sites. A study of headwater streams in Kentucky (USA) found that unmined reference streams ranged from 15 to 73 % Ephemeroptera and <6 % in mined streams (Pond 2010). A similar result was obtained in a study of an

AMD polluted Irish river (River Avoca) which reported that the upstream reference site had 43.8 % Ephemeroptera compared to 5.1 % at the AMD affected site (Gray and Delaney 2008).

The current results from the Bargo River also differed to two other studies that assessed the impact of coal mine waste discharges to rivers in the nearby Blue Mountains area of the Sydney Basin (Grose River; Wright and Burgin 2009a; Wollangambe River, Belmer et al. 2014). Coal mine wastes, in both the Grose River and Wollangambe River, caused the two biotic indices (taxonomic richness and abundance) of stream macroinvertebrates to sharply (and significantly) fall below both mine waste discharges. Similar to the current study, % EPT did not change significantly below the coal mine waste discharge in the Wollangambe River (Belmer et al. 2014). Coupled with the results of this study, it is a possible explanation that % EPT is not a sensitive indicator of ecological impairment from coal mining waste discharges. A similar conclusion was reached following an investigation of heavy metal pollution in the Arkansas River in Colorado which also reported that the EPT index was not effective at detecting ecological impairment from mild levels of metal contamination (Kiffney and Clements 1994).

Depleted levels of dissolved oxygen were frequently recorded in Redbank Creek. The Bargo River generally maintained adequate levels of DO for aquatic ecosystems (>85 % saturation; ANZECC 2000) at both sites, although there was a slight reduction of DO below the mine discharge. All sites on Redbank Creek recorded very low DO levels with all three sites recording DO minimums below 10 % saturation. The lowest mean level of 21.9 % DO was obtained at RB3. Other investigations have reported a link between coal mining and lowering of dissolved oxygen in surface waters. For example, Redstone Creek in Pennsylvania (USA) was heavily impacted by AMD from coal mine wastes and reported dissolved oxygen at levels stressful for aquatic ecosystems (Moon and Lucostic 1979). Other studies did not detect any connection between coal mining and dissolved oxygen. For example, a study of multiple streams subject to differing degrees of metal pollution in the Colorado Mountains found that dissolved oxygen levels were similar at all classes of metal pollution compared to unmined reference waterways (Clements et al. 2000). Jankowski (2007) observed that lower dissolved oxygen levels were recorded, along with higher metal concentrations, in the vicinity of longwall

subsidence compared to non-mine-affected upstream reaches. Similarly, the subsidence affected abandoned Majestic Mine in Ohio (USA) released mine drainage with very low dissolved oxygen levels, particularly in seasonal high flow conditions (Pigati and López 1999).

Both coal mining activities (waste disposal and subsidence) in this study increased surface water EC. In both cases, the EC increased from background (upstream) levels (<250 $\mu\text{S}/\text{cm}$) to higher levels downstream, well above recommended water Australian quality guidelines for ecosystems (350 $\mu\text{S}/\text{cm}$; ANZECC 2000). The waste discharge increased mean EC in the Bargo River by nearly five times, from 186 to 1078 $\mu\text{S}/\text{cm}$, a level just over three times higher than guidelines for protection of aquatic ecosystems (ANZECC 2000). The longwall subsidence increased mean EC of Redbank Creek from 247 to 1195 $\mu\text{S}/\text{cm}$. There is considerable comparative data in the literature on the EC effects of coal mine waste discharges on surface waters. These findings show that the magnitude of the EC impact of this coal mine is approximately mid-range compared to other international coal mine studies. On the lower end of the scale, a study from north-western Spain reported background EC of non-mined streams of 18–34 $\mu\text{S}/\text{cm}$ increased more than fivefold to 202–472 $\mu\text{S}/\text{cm}$ below the coal mines (García-Criado et al. 1999). In the nearby Blue Mountains, the EC of the Grose River rose to a modest degree, from 83 to 151 $\mu\text{S}/\text{cm}$, due to inflow of a small tributary containing mine drainage from a disused mine (Wright and Burgin 2009a). An investigation of West Virginia (USA) streams in coal mined areas averaged 1023 $\mu\text{S}/\text{cm}$ compared to 62 $\mu\text{S}/\text{cm}$ in unmined streams (Pond et al. 2008). Representing the higher end of the EC scale were four coal mines in Britain with EC levels in their waste discharges of 2850 to 5050 $\mu\text{S}/\text{cm}$ (Banks et al. 1997). Other coal mines in the Sydney basin also increase surface water EC, but to a lesser level than the current study. The Georges River EC increased from 175 $\mu\text{S}/\text{cm}$ upstream of the Westcliff mine waste discharge to 710 $\mu\text{S}/\text{cm}$ downstream (Wright 2012). Battaglia et al. (2005) reported EC in a stream in the western Blue Mountains stream (Neubecks Creek) impacted by coal mining had a mean level of EC of 787 $\mu\text{S}/\text{cm}$ compared to two reference sites (132 and 17 $\mu\text{S}/\text{cm}$).

There are fewer studies examining the impact of subsidence from underground coal mining to changes in surface water EC. One of the only similar studies, where longwall subsidence fractured bedrock and

reduced stream flow, which re-emerged in the stream channel further downstream, was by Jankowski (2007). This study was conducted on a longwall subsidence damaged stream, the Waratah Rivulet, in the in the Woronora Plateau south of Sydney (Jankowski 2007). It reported similar background EC (200–280 $\mu\text{S}/\text{cm}$) to the current study, above the subsidence area, to mildly elevated levels (260–340 $\mu\text{S}/\text{cm}$) in the vicinity of the subsidence. In comparison, the surface water of the current study had a much higher level of EC. Booth and Bertsch (1999) investigated the impact of longwall coal mining on groundwater chemistry and concluded that some aquifers varied from site to site and resulted in negligible to major increases in groundwater EC, largely dependant on the groundwater and geological conditions of each site.

The two mining activities in this study were associated with contrasting impacts on surface water pH. The pH of the Bargo River, upstream of the waste, was mildly acidic with a mean pH of 6.54, and the site below the coal mine was strongly alkaline with a mean of 8.19. A contrasting change in pH was associated with the subsidence in Redbank Creek which decreased from an upstream mean pH of 6.79 dropping to a mean pH of 5.97. In comparison, other international coal mine discharges have reflected stronger acid mine drainage effects and associated river acidification. Very low pH (ranging from 2.6 to 4.2) was reported from ten streams in New Zealand affected by abandoned coal mines (Winterbourn 1998). Similarly, two streams containing coal mine effluent in the Hocking River (Ohio, USA) had mean pH levels of 3.0 and 3.1 (Verb and Vis 2000) compared to unmined reference streams with mean pH levels of 7.1 and 7.2 (Verb and Vis 2000). Streams in the vicinity of the Green Valley mine (Indiana, USA) with alkaline unmined reference streams (pH 7.6 to 8.6) and strongly acidic coal mine effluent with pH levels ranging from 2.2 to 4.6 (Brake et al. 2001). There is very little information in the literature on pH changes in surface waters subject to longwall mine subsidence. The subsidence study of Waratah Rivulet by Jankowski (2007) recorded an alkalization of surface waters with upstream pH (6.5–7.1) increasing to 7.7 in the area affected by subsidence. The Clarence coal mine in the Blue Mountains area of the Sydney Basin also increased in surface water pH (Belmer et al. 2014). The mine wastewater discharges lifted the Wollangambe River pH from 5.6, above the mine waste discharge, to approximately pH of 8 below the mine, attributed to

lime dosing treatment of the mine waste by the coal mine (Belmer et al. 2014).

The coal mine operation modified the ionic composition of surface waters in different ways. The background water chemistry above the waste discharge and the subsidence both showed that both waterways were dominated by sodium cations and chloride anions. Studies of streams in naturally vegetated catchments across the Sydney Basin have also reported the dominance of sodium and chloride ions (e.g. Tippler et al. 2014). This is also broadly typical of many coastal flowing streams in south-eastern Australia with the ionic composition of streams and rivers dominated by sodium and chloride ions (Hart and McKelvie 1986). Sodium remained the dominant cation in the Bargo River, below the mine waste, but the order of the cation concentrations in the Bargo River changed from $\text{Na} > \text{Mg} > \text{Ca} > \text{K}$ above the mine to $\text{Na} > \text{K} > \text{Ca} > \text{Mg}$ downstream. The mine waste discharge changed the anion dominance from $\text{Cl} > \text{HCO}_3 > \text{SO}_4$ in the Bargo River above the discharge to $\text{HCO}_3 > \text{Cl} > \text{CO}_3 > \text{SO}_4$. This was the only sampling site in the study where carbonate was detected. In Redbank Creek, the cation dominance changed from $\text{Na} > \text{Ca} > \text{Mg} > \text{K}$ upstream to $\text{Na} > \text{Mg} > \text{Ca} > \text{K}$ downstream. At Redbank Creek, the anion dominance was unchanged from the background ($\text{Cl} > \text{HCO}_3 > \text{SO}_4$). Jankowski (2007) reported that longwall subsidence caused calcite dissolution and changed the ionic composition of Waratah Rivulet from $\text{Na} > \text{Ca}$ and $\text{Cl} > \text{HCO}_3$ above the subsidence to $\text{Ca} > \text{Na}$ and $\text{HCO}_3 > \text{Cl}$ dominance downstream. A detailed study of the abandoned Green Valley coal mine in USA (Brake et al. 2001) reported no change in dominance of cations due to the influence of the mine drainage, but that anion dominance shifted from $\text{HCO}_3 > \text{SO}_4 > \text{Cl}$ in uncontaminated tributaries to $\text{SO}_4 > \text{Cl} > \text{HCO}_3$ below the mine.

The sulfate levels in surface waters in this study were very low by comparison with other coal mine investigations. The sulfate concentration only increased as a result of the mine waste discharge into the Bargo River, from a mean of 4.6 mg/L upstream to 9.7 mg/L downstream. The sulfate levels fell downstream of the subsidence in Redbank Creek to < 6 mg/L. In comparison, a study of coal mining and stream water quality in the Lower Cheat River basin in West Virginia (USA) recorded higher levels of sulfate in unmined streams (mean 14 mg/L) than any site in the current study (Petty et al. 2010). The West Virginia study also reported mean sulfate levels in one area of low mining

intensity of 28 and 338 mg/L in an area of high mining intensity (Petty et al. 2010). Even higher sulfate levels were reported from coal mine wastes in Britain by Banks et al. (1997) where three pumped mines in Durham had sulfate concentrations ranging from 380 to 1170 mg/L. A range of moderate to very high levels of sulfate (67.7–5240 mg/L) were found in streams contaminated by the Green Valley coal mine in Indiana, USA (Brake et al. 2001).

Both the waste discharge and the longwall subsidence increased the concentration of several metals in surface waters. Of most ecological concern were the metals nickel and zinc. Both were at concentrations higher than recommended for Australian freshwater ecosystems (ANZECC 2000). The mean zinc concentration increased from a mean of 7.9 µg/L (upstream) to 38.6 µg/L below the waste discharge into the Bargo River. The mean zinc concentration increased from 15.4 to 86.3 µg/L in Redbank Creek below the subsidence zone (RB3). The nickel concentrations in Bargo River increased from 1.65 µg/L above the mine to 35.6 µg/L downstream of the waste. In Redbank Creek, the upstream concentration of nickel increased from <1 µg/L above the subsidence to 4.5 and 30.5 µg/L at the two subsidence affected sites. The levels of zinc and nickel were higher than was reported at coal mine affected streams in West Virginia which had slightly higher nickel in mined streams (mean 14.2 µg/L) compared to unmined streams (<10 µg/L) and no elevation in zinc levels in mine affected streams (Pond et al. 2008). Much higher zinc concentrations (at unmined reference streams) of 58 and 70 µg/L in the River Don (Lancashire, UK) increased to 372 µg/L in sections of the river receiving coal waste (Greenfield and Ireland 1978). The enrichment of zinc and nickel concentrations in the current study were modest when compared to a study of water chemistry below the Green Valley coal mine (Indiana USA) with zinc often above 5000 µg/L and nickel levels above 500 µg/L and as high as 3780 µg/L (Brake et al. 2001). In the Blue Mountains (about 80 km from the current study), a disused coal mine (Canyon Colliery), about 10 km to the south-east of the Clarence coal mine, discharged much higher levels of zinc at mean levels of 595 µg/L into the Grose River lifting zinc concentrations from <5 to 388 µg/L (Wright and Burgin 2009a).

Two other metals (iron and manganese) were most elevated at RB3, the most strongly subsidence modified site on Redbank Creek. This site recorded mean iron concentrations of just over 20 mg/L and manganese concentrations of almost 5 mg/L. In comparison, the study of

Waratah Rivulet (Jankowski 2007) reported maximum concentrations of iron of up to 12 mg/L and manganese concentrations of nearly 1 mg/L. Although not attributed to subsidence, coal mine drainage from waterways below the abandoned Green Valley had higher iron and manganese concentrations, with levels of iron up to 273 mg/L and manganese of 5.7 mg/L (Brake et al. 2001).

This study constitutes one of the first investigations of water quality and macroinvertebrate ecosystem response to two forms of water quality impairment from a single underground coal mine. The results show that aquatic ecosystems can be impacted from the disposal of coal mine wastes and also from longwall subsidence. The water quality impacts of each coal mine disturbance are complex and different water chemical changes were associated with each forms of disturbance. Ecotoxicological testing is required to determine the most influential physical and chemical factors that are responsible for the ecological impacts observed in the macroinvertebrate communities. The candidates for causing most ecological stress in this study include: depleted dissolved oxygen, changes in pH, increased EC, modified ionic composition and increased metal concentration (zinc, aluminium, manganese and nickel).

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