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Coal Mine Water Pollution and Ecological Impairment of One of Australia's Most 'Protected' High Conservation-Value Rivers

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Abstract The environmental regulation of a coal mine in the greater Sydney area has failed to recognise the importance of and protect a high conservation-value river located in a World Heritage listed area. This study measured the water quality and ecological health (using macroinvertebrates) of the Wollangambe River and its tributaries near the point of the waste water discharge of a coal mine and assessed the longitudinal impact for 22 km downstream. The investigation revealed two important aspects. The first is the significant impact of the waste water discharge when compared to the otherwise near-pristine condition of the high conservation-value river system. The second is the spatial extent of the pollution from the mine that extends at least 22 km downstream from the outflow of coal mine wastes. The resulting water pollution is causing major impairment of the aquatic ecosystem, with reduced abundance, taxonomic richness and loss of pollution-sensitive macroinvertebrate groups. Water pollution from the mine includes thermal pollution, increased salinity and increased concentrations of zinc and nickel. The mine's waste discharge also strongly modified the river's ionic composition. The study also highlights the failure of the regulatory and governance systems that enable the mine to operate in a manner that causes major environmental impacts.

Keywords World heritage · Environmental management · Freshwater ecosystems · Water quality · Macroinvertebrates · Ionic composition

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1 Introduction

Underground coal mining has been a major industry in the Blue Mountains region, to the west of Sydney, for more than a century (Cremin 1989; Macqueen 1997). It has supported the development of many towns in the region and influenced the region's industry, including two major coal-fired power stations (Cremin 1989). There are several active and disused coal mines that often have ongoing impacts on local streams and rivers located within the western Blue Mountains region (Macqueen 1997; Birch et al. 2001; Harrison et al. 2003; Battaglia et al. 2005; Wright and Burgin 2009a, b; Wright and Ryan 2016).

It is well documented that coal mining activity can cause significant water pollution impacts globally, including the USA (Verb and Vis 2000; Brake et al. 2001; Pond et al. 2008; Griffith et al. 2012), Brazil (Lattuada et al. 2009), the UK (Jarvis and Younger 1997; Younger 2004; Johnson 2003; 2004) and New Zealand (Winterbourn 1998; Gray and Harding 2012). This is often associated with acid mine drainage (AMD) and modified stream pH (Banks et al. 1997; Verb and Vis

2000; Brake et al. 2001), elevated salinity (García-Criado et al. 1999; Pond et al. 2008) and elevated heavy metals (Brake et al. 2001; Johnson 2003; Pond et al. 2008). These and other water quality impacts are evident from various former and current mines in the Blue Mountains region (Birch et al. 2001; Harrison et al. 2003; Battaglia et al. 2005; Wright and Burgin 2009a, b; Wright 2012; Price and Wright 2016). The connection between coal mine waste water, water pollution and aquatic ecological impacts in waterways is also widely reported (e.g. Scullion and Edwards 1980; Jarvis and Younger 1997; Winterbourn 1998; García-Criado et al. 1999; Pond et al. 2008; Gray and Harding 2012; Griffith et al. 2012). However, there are few studies that investigate water pollution from coal mines to waterways in otherwise largely undisturbed areas of high conservation-value such as a World Heritage site (e.g. Wright and Burgin 2009a, b; Weis 2014; Price and Wright 2016). This may be due to the paucity of mines in highly significant environmental areas or rather that independent research is yet to focus on the impact of mines in such settings.

Within the Blue Mountains and the associated World Heritage area, previous investigations of water pollution from coal mines have focused on a number of disused mines. Battaglia et al. (2005) reported that AMD from untreated wastes from historic coal mine disturbance caused low pH (5.1) and elevated zinc (c. 500 μg/L) and nickel (c. 400 μg/L) and degradation of stream macroinvertebrate assemblages within Neubecks Creek, Lithgow. More recently, Wright and Burgin (2009a, b) and Price and Wright (2016) reported zinc and nickel concentrations well above ecological guideline limits were discharged from a closed mine into a highly sensitive stream (Dalpura Creek) at the headwaters of the Grose River catchment. Wright and Burgin (2009a, b) also reported ecological impairment for many kilometres downstream of the discharge evident through reduced invertebrate taxonomic richness, abundance and reduction of pollution sensitive groups, such as EPT families (Lenat 1988) and chironomid species richness.

This study investigates water pollution and ecological impairment resulting from a currently active underground coal mine located in close proximity to, and discharging waste water to a river flowing into and within, the Greater Blue Mountains World Heritage Area (GBMWHA). This mine was approved prior to the listing of the Greater Blue Mountains World Heritage Area (UNESCO 2009). The environmental impacts from coal mining activity were included in the nomination for the Blue Mountains area to be listed as a World Heritage Area (Australian Government 1998) and this highlighted the important role of the New South Wales (NSW) state government Environment Protection Authority (EPA) in regulating coal mines and so doing protecting the World Heritage Area (Australian Government 1998).

This study conducted an assessment of the environmental impacts of the Clarence Colliery on the Wollangambe River through quantitative macroinvertebrate surveys and temporally replicated water quality sampling. The study was designed to measure any adverse impacts on freshwater ecosystems downstream from the mine waste water discharge. This investigation was framed around answering three questions:

1. How do stream macroinvertebrates respond to coal mine waste water discharges?
2. What are the key water quality changes (particularly geochemistry and metals) to the stream downstream of the mine discharge?
3. Does the stream show any downstream signs of ‘recovery’ from pollution effects?

The study design anticipated an impact downstream of the discharge point. However, the focus included an investigation of the severity and spatial extent of the water chemical and ecological impact given the environmental significance of the area and receiving water (a World Heritage site and state listed river). The outcome of the research was also positioned to inform future development assessment and ongoing pollution licencing by environmental regulators to achieve greater protection for important and highly valued aquatic environments.

2 Methods

2.1 Study Area

This study investigates the impacts arising from the waste water discharge from the Clarence Colliery into the Wollangambe River. This underground coal mine is located approximately 8.5 km east of Lithgow, in south-eastern Australia. It was approved by the NSW Government on 15 June 1976 and was granted an extension by the Minister for Planning to its mining activity in 2005. The mine has operated continuously since 1980 (Cohen 2002). Waste water is generated through a combination of groundwater seeping into the mine and becoming contaminated from underground coal mining operations (Cohen 2002). Additionally, mine water is also contaminated through the coal washing process and related runoff from surface mining activities (Cohen 2002). The surface operations of the mine and underground workings are located on a combination of freehold land and NSW State Forest land under mining leases.

The discharge of waste water is regulated by the NSW EPA under the Environmental Pollution Licence (EPL) 726 held for the mine (Graham and Wright 2012). All details for this licence are freely available (EPA 2016). This licence specifies volumes of waste water and 18 individual physical or chemical attributes (generally as concentrations) comprising chloride, fluoride, oil and grease, pH, total suspended solids, iron, manganese, arsenic, boron, cadmium, copper, lead, mercury, selenium, silver, sulphate, chromium and fluoride. The mine continuously discharges waste water to the Wollangambe River via a small unnamed tributary through a point labelled as 'LDP002' in the mine's EPL 726 licence (EPA 2016). The inflow of the mine wastes to the Wollangambe River occurs a short distance of approximately 1.3 km upstream of the boundary of Blue Mountains National Park and then further downstream the Wollangambe River flows from Blue Mountains National Park into Wollemi National Park (Fig. 1).

Beyond the boundaries of the mine, the majority of the study area falls within Blue Mountains National Park and Wollemi National Park. Wollemi National Park is the second largest National Park in NSW with an area of 488,620 ha (NP&WS 2001b) and together with Blue Mountains National Park forms part of the Greater Blue Mountains World Heritage Area (GBMWHA). There is no urban development within the study catchment with other land uses confined to an open-cut sand mine (adjacent to the mine) and State Forest that operates some softwood plantations and native woodlands with associated unsealed access tracks.

2.2 Environmental Significance of the Study Area

The environmental and conservation significance of this region and its waterways are recognised by four specific legislative processes. Blue Mountains and Wollemi National Parks are established under the *National Parks and Wildlife Service* Act 1974 (NSW) and both are managed according to the Blue Mountains National Park Plan of Management (NP&WS 2001a) and Wollemi National Park Plan of Management (NP&WS 2001b). The conservation significance of the rivers and streams in the Wollemi National Park, which includes the Wollangambe River, was declared a 'Wild River' (NSW OEH 2015b) in 2008 under the NSW *National Parks and Wildlife* Act (1974). The Wollangambe catchment, which includes the majority of the study area, was declared Wilderness Area in 1991 under *Wilderness* Act 1987 (NSW). This declaration recognises that most of the area is undisturbed with essentially intact plant and animal communities in a pristine condition unmodified by human activity. In 2000, the Blue Mountains and Wollemi National Parks, including most of the study area, and five other national parks in the region were declared a World Heritage Area under the UNESCO and World Heritage Convention (identification number 917). One of the UNESCO criteria for this World Heritage listing (UNESCO 2009) is indicative of the conservation significance of the terrestrial and aquatic ecosystems of the GBMWHA as it must contain 'outstanding examples representing significant on-going ecological and biological processes in the evolution and development of terrestrial, fresh water, coastal and marine ecosystems and communities of plants and animals'. A World Heritage Conservation listing requires the Federal Environment Minister to consider and give approval for any action or proposed action that is likely to have an adverse impact on the World Heritage values (*Environment Protection and Biodiversity Conservation* Act 1999 (*Cth*)). For the purpose of the Clarence Colliery, this may include a change to its operation that may result in a substantial increase in concentrations of suspended sediments, nutrients, heavy metals, hydrocarbons or other pollutants (Australian Commonwealth Government 2013).

2.3 Study Design and Sampling Sites

This study was conducted within the upper Wollangambe River catchment. The relatively small area used for the study was chosen to increase the likelihood of the streams having a similar geochemistry and a similar regional 'pool' of macroinvertebrate species (Corkum 1989). To fully assess the impact of the mine, it would have been preferable to measures changes over a longer time period

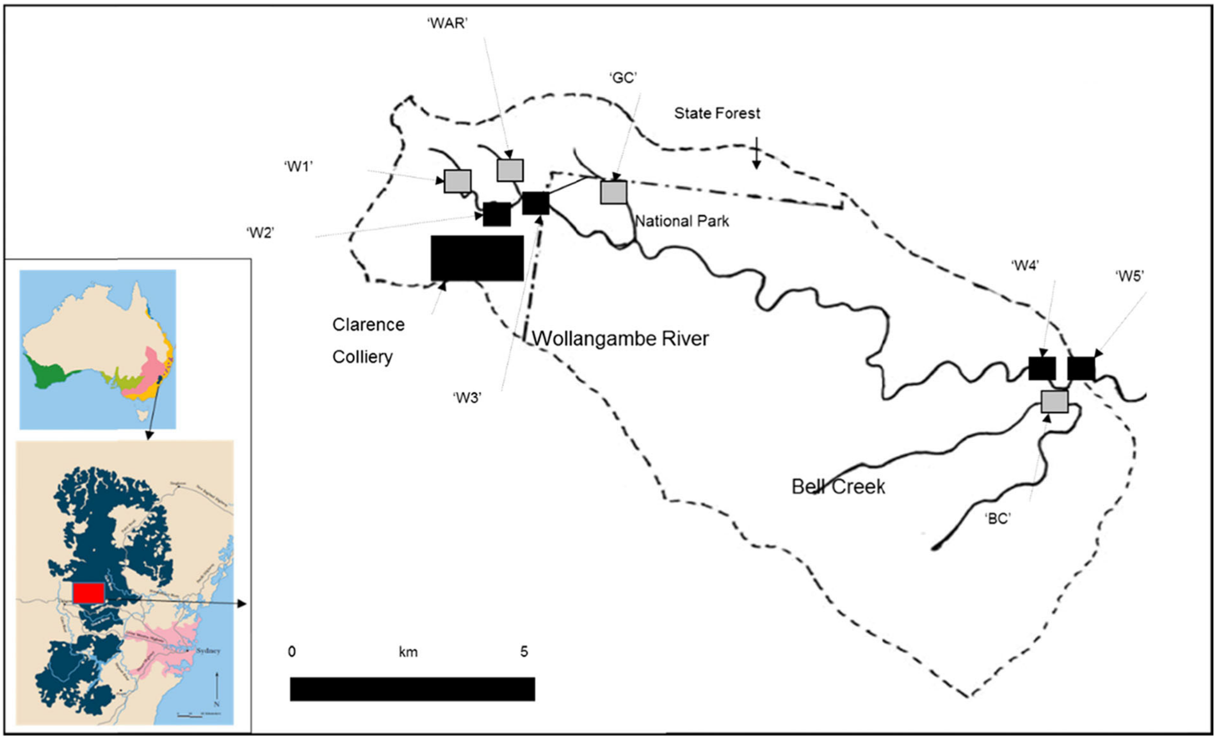


Fig. 1 Map showing location of the study area (*inset* location within Australia and location within the Greater Blue Mountains World Heritage Area). The Clarence Colliery is shown and the sampling sites are shown as reference sites (*grey markers*) and sites downstream of the mine (*black markers*). The tenure of the land is marked as State Forest and National Park. The main waterways, Wollangambe River and Bell Creek, are labelled

starting prior to the construction of the mine using a 'before versus after, control versus impact' (BACI) design (see Underwood 1991). This was not possible as the coal mine began operations about 32 years previously and no comprehensive data was collected prior to its operation (Cohen 2002). Water quality and stream invertebrate samples were collected from multiple reference sites to enable comparison with physically similar sites (size, altitude, geology) exposed to the coal mine waste water (Fairweather 1990).

Sampling was carried out on four upland streams (Table 1) across eight sampling locations within the upper Wollangambe River catchment (11,138 ha) in the Greater Blue Mountains area (33° 28′ S, 150° 17′ E) (Fig. 1). Four sites located along the Wollangambe River were used to sample the impacts of the mine below the licenced discharge point. These were grouped into two pairs. The first pair (W2 and W3) was situated within 1.2 km of the mine waste discharge point. This sampling pair is referred to as *Wollongambe State Forest* (WSF) being located in NSW State Forest land immediately upstream of the National Park Boundary. The second pair (W4 and W5) was located 22 km below the mine discharge point. These are referred to as *Wollangambe National Park* (WNP) and represent the river within the conservation area (National Park, Wilderness Area, Wild River, World Heritage Area). The distance between WSF and WNP reflects the rugged and inaccessible terrain along this stretch of the Wollangambe River. The sampling design included an equal sampling effort at sites downstream of the mine and at 'reference' sites unaffected by coal mine wastes. The four reference sites included one site upstream of the mine discharge at the headwaters of Wollangambe River (W1) and three from tributaries of the Wollangambe River (Fig. 1, Table 1). Sampling sites exposed to mine waste and unaffected reference sites were selected to represent a comparable spatial scale and altitudinal profile (992 to 741 m ASL; Fig. 1, Table 1). Bell Creek (Fig. 1), in particular, drained a similar subcatchment to the upper Wollangambe River and was considered to have similar size and flow characteristics to the Wollangambe River downstream of the coal mine waste discharge.

Table 1 Summary details for each of the sampling sites (and sampling categories: REF, WSF, WNP) in the current study

Site name (sampling category)	Site code	Co-ordinates	Width	Vegetation (Keith and Benson 1988)	Stream order	Altitude (m ASL)
Wollangambe River 200 m above coal mine discharge (REF)	W1	33° 27′ 24.3″ S, 150° 14′ 58.19″ E	0.5–1 m	Hanging swamp/Blue Mountains Riparian Complex	1st	992
Wollangambe River 100 m below coal mine discharge (WSF)	W2	33° 27′ 21.27″ S, 150° 15′ 06.73″ E	1–2 m	Tall open forest form/Blue Mountains Riparian Complex	1st	979
Wollangambe River 1.2 km below coal mine discharge (WSF)	W3	33° 27′ 20.37″ S, 150° 15′ 27.43″ E	1.5–2 m	Tall open forest form/Blue Mountains Riparian Complex	2nd	956
Wollangambe River 21.0 km below coal mine discharge (WNP)	W4	33° 29′ 21.07″ S, 150° 21′ 13.74″ E	2–3 m	Sandstone warm temperate rainforest/Blue Mountains Riparian Complex	2nd	749
Wollangambe River 22.0 km below coal mine discharge (WNP)	W5	33° 29′ 15.18″ S, 150° 21′ 20.98″ E	4–5 m	Sandstone warm temperate rainforest/Blue Mountains Riparian Complex	3rd	741
Unnamed Creek ('Waratah Ck') (REF)	War	33° 27′ 16.57″ S, 150° 15′ 26.99″ E	0.5–1 m	Closed forest form/Tableland Riparian Scrub Complex	1st	958
Unnamed creek above Gooches Crater (REF)	GC	33° 27′ 08.76″ S, 150° 15′ 48.04″ E	0.5–1 m	Tall open forest form/Sheltered Gully Brown Barrel Ferny Forest	1st	967
Bell Creek (REF)	BC	33° 29′ 23.97″ S, 150° 21′ 14.84″ E	1–2 m	Sandstone warm temperate Rainforest/Blue Mountains Riparian Complex	2nd	748

Aquatic macroinvertebrates were collected to measure any freshwater ecosystem changes associated with the coal mine waste discharges. Macroinvertebrate samples were collected from all sites in May (Austral autumn) 2013 at each site. Five quantitative benthic macroinvertebrate samples were collected from cobble riffle habitats which were widely available at all sampling locations (Resh and Jackson 1993; Wright et al. 1995). 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). Macroinvertebrate samples were collected using a 'kick sampling' technique. A 'kick' net with a frame of 30 × 30 cm square aperture with a robust 250-μm mesh was used (Rosenberg and Resh 1993; Wright and Burgin 2009a). Sampling was conducted by disturbing the stream benthos for 30 s over a 900-cm^2 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 a 70% ethanol solution.

In the laboratory, the entire collected sample was examined and sorted under a high-resolution dissecting microscope (×10 to ×60) to extract the macroinvertebrates from stream detritus (e.g. leaves, sticks, rocks, gravel). Macroinvertebrate identification was conducted using the identification keys recommended by Hawking (2000). All insect groups were identified to the family taxonomic level as this approach has been demonstrated to provide adequate taxonomic resolution for impact assessment (Wright et al. 1995; Wright and Ryan 2016). Two non-insect groups (Oligochaeta, Hydracarina) were not identified to the family level due to identification difficulties.

Water chemistry was investigated on four occasions (December 2012; early and late February 2013 and May 2013). The sampling period had higher rainfall than normal and no water quality sampling was undertaken for at least 48 h following any occurrences of heavy rain (20 mm/day). We recognised the importance of undertaking macroinvertebrate sampling in typical dry weather flow conditions (e.g. Chessman 1995). The area received below average rainfall in April and May 2013 and river and stream flows were assessed as being representative of dry weather, before sampling of macroinvertebrates was conducted in May 2013.

Physiochemical water quality attributes including pH, electrical conductivity (EC), dissolved oxygen (DO) and water temperature were measured in situ using a TPS AQUA-Cond-pH meter (for pH and electrical conductivity) and a YSI ProODO meter (DO and water temperature). The calibration of each meter was checked on each day and adjusted if necessary. Water samples were collected in clean and unused sample containers provided by a commercial testing laboratory. Samples were chilled and delivered to the laboratory for analysis. All water 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 and aluminium). Analytical QA/QC procedures within the testing laboratory included the use of sample blanks and spiked samples to ensure the reliability of analytical procedures.

2.4 Biotic Indices and Data Analysis

Macroinvertebrate family richness, total abundance and EPT (Ephemeroptera, Plecoptera and Trichoptera) abundance and taxonomic richness biotic indices were used to help detect and measure pollution related changes to the Wollangambe River ecosystem. Macroinvertebrate abundance was included, although it is often not assessed due to the qualitative nature of many widely used rapid assessment methodologies (e.g. Resh and Jackson 1993; Chessman 1995). However, abundance was shown to be useful in previous coal mine impact studies (Clements et al. 2000; Wright and Burgin 2009a). Similarly, EPT biotic indices were also calculated due to their successful use in previous mine impact studies (Clements et al. 2000; Pond et al. 2008; Merriam et al. 2011). The EPT index is based on the relative abundance or taxonomic richness of three common macroinvertebrate orders that have demonstrated sensitivity to disturbance and degraded water quality (Lenat 1988; Lenat and Penrose 1996). The EPT index is widely reported as a robust and effective index for measuring impairment to stream macroinvertebrates (e.g. Plafkin et al. 1989; Barbour et al. 1992; Hickey and Clements 1998; Metzeling et al. 2006) and is one of the most effective and relatively simple biotic index for evaluating ecological impairment from coal mine water pollution (Pond et al. 2008; Wright and Burgin 2009a; Wright et al. 2015). The use of stream ecosystem data is not currently included in the current monitoring programme or pollution licence of the coal mine operators or routinely undertaken by the environmental regulator (EPL 726; EPA 2016).

Multivariate analyses of macroinvertebrate community data have been demonstrated to be an effective statistical technique to evaluate the ecological response of freshwater macroinvertebrates to disturbance and pollution from coal mining (Merriam et al. 2011; Pond et al. 2008; Wright et al. 2015). Non-metric multidimensional scaling (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). Two-dimensional ordination plots represented the dissimilarity among samples. All reference site samples were grouped to test differences by a two-way analysis of similarity (ANOSIM; Clarke 1993) between reference sites and sites at the two categories WSF and WNP downstream of the waste discharges. The sites downstream from the mine discharge point were grouped as WSF and WNP (Section 2.3). These multivariate analyses were achieved using the software package PRIMER version 5 (Clarke 1993).

All water chemistry data and macroinvertebrate data biotic index data was checked for homogeneity of variances (using Levene's test) and normality of distribution and was found to generally be skewed. The non-parametric Kruskal-Wallis tests were then performed with data grouped according to different sampling locations (reference sites, WSF and WNP). The univariate statistical analysis was performed using IBM SPSS Statistics version 21.

3 Results

3.1 Macroinvertebrates

A total of 1156 macroinvertebrates from 33 taxa (mostly families) were collected, the majority being insects (Table 2). Reference sites accounted for 85.8% of all invertebrates collected. Family richness varied highly significantly ($p = 0.0001$) according to sampling site category (Table 2). Family richness was highest at the reference sites (mean 8.7) with a total of 30 of the 33 taxa detected at reference sites (Tables 2 and 3, Fig. 2). Family richness was lowest immediately downstream of the mine discharge at WSF sites (mean 3.2 families) with a total of 8 of the 33 taxa detected (Tables 2 and 3,

Table 2 Summary statistics of macroinvertebrate (macro) and water physical and chemical data indicating the *p* value, range, mean (and median) for each variable, according to sampling site category (reference sites vs. WSF sites vs. WNP sites) (see Table 1 for site details)

Macro. variables	*p* value (K-W)	Reference sites (REF)		0.2–1.2 km below mine (WSF)		22 km below mine (WNP)	
		Range	Mean (Med.)	Range	Mean (Med.)	Range	Mean (Med.)
Family richness	0.0001	3–14	8.7 (9)	3–4	3.2 (3)	2–10	4.8 (4)
Abundance	0.0001	8–166	50.7 (38.5)	3–17	6.1 (4)	3–34	10.7 (8)
EPT abundance	0.0001	0–87	23.9 (20)	0–3	0.8 (0.5)	2–23	5.9 (3.5)
EPT richness	0.0001	0–8	4.1 (4)	0–2	0.6 (0.5)	1–4	1.8 (1.5)
EC (μS/cm)	0.0001	15–36	25.8 (26)	252–503	386.5 (411.5)	182–326	246.3 (242)
pH (pH units)	0.0001	4.9–6.3	5.48 (5.39)	6.9–8.7	7.54 (7.44)	6.4–6.94	6.65 (6.67)
Water temp. (°C)	0.0001	7.7–15.6	12.59 (13.7)	12.7–17.9	16.47 (17.6)	9.4–18.8	15.1 (17.05)
DO (% sat.)	0.0001	14.8–99.1	72.4 (82.0)	72.2–105.9	95.3 (99.3)	92.4–124.1	105.0 (101.9)
Hardness (mg/L)	0.0001	Bd.	Bd.	163–223	193.2 (196.5)	35–126	86.1 (92.0)
Bicarbonate (mg/L)	0.0001	Bd. - 1	0.54 (Bd.)	12–22	19.7 (21)	2–8	5.58 (6)
Sulphate (mg/L)	0.0001	Bd. - 7	1.6 (1)	151–201	177.8 (181)	34–122	83.4 (87)
Chloride (mg/L)	0.0001	4–8	5.37 (5)	4	4 (4)	4–6	5.17 (5)
Sodium (mg/L)	0.228	2–4	3.3 (3.0)	3–4	3.7 (4.0)	3–4	3.5 (3.5)
Calcium (mg/L)	0.0001	Bd.	Bd.	47–63	55.0 (54.5)	9–36	24.4 (25.5
Magnesium (mg/L)	0.0001	Bd.	Bd.	11–16	13.58 (13.5)	3–10	6.08 (6.5)
Potassium (mg/L)	0.0001	Bd.	Bd.	4–5	4.25 (4)	1–3	1.83 (2)
Zinc (μg/L)	0.0001	Bd. - 18	4.02 (Bd.)	94–180	124.9 (123.5)	40–64	50.2 (48.5)
Nickel (μg/L)	0.0001	Bd. - 2	0.69 (Bd.)	78–141	105 (98.5)	36–48	41.2 (40)
Aluminium (μg/L)	0.0001	60–880	200.6 (145)	70–320	26.7 (20)	0–90	46.7 (50)

Bd. below detection limits

Fig. 2). Family richness in the conservation area (22 km) downstream of the mine discharge at the WNP sites was about 45% lower (mean 4.8 families) than the mean of reference sites with a total of 16 of 33 taxa detected (Tables 2 and 3, Fig. 2).

Macroinvertebrate abundance varied highly significantly ($p = 0.0001$) according to sampling site category (Table 2). Macroinvertebrate abundance was highest at reference sites with a mean of 50.7 invertebrates per replicate (Table 2, Fig. 2). Mean abundance in the Wollangambe River below the mine (WSF) was 88% lower than at references sites (mean of 6.1 invertebrates per replicate; Table 2). Further downstream of the mine discharge point (at WNP), the mean abundance was 10.7 invertebrates per replicate, 79% lower than the mean abundance at reference sites (Table 2, Fig. 2).

EPT richness and EPT abundance both varied significantly ($p = 0.0001$) according to sampling site category (Table 2). At the reference sites EPT richness (mean 4.1) and EPT abundance (23.9) had the highest scores. Immediately below the mine discharge (WSF), EPT richness (mean 0.6) and abundance (0.8) reported the lowest scores (Fig. 2). At the most downstream mine affected sites, and within the World Heritage Conservation Area (WNP), EPT richness (mean 1.8) and abundance (5.9) recovered only slightly (Table 2, Fig. 2). All EPT orders had reduced abundance below the mine, compared to reference sites. No mayflies (Ephemeroptera), a sensitive order of invertebrates, were collected from the river at the two sites immediately below the mine (WSF; Table 2).

Multivariate analysis revealed that the macroinvertebrate communities were highly impaired in the Wollangambe River below the mine, compared to those collected from reference sites. The nMDS ordination showed that sites immediately below the mine (WSF) and those further downstream (WNP) clustered separately from reference sites, with minor overlap of samples (Fig. 3). The nMDS stress value (0.18) indicated that, in two dimensions, the nMDS was a good to fair representation of the original data (Clarke 1993). Analysis by ANOSIM confirmed that macroinvertebrate

Table 3 Macroinvertebrate family presence (positive detection marked by X) from the three sampling site categories. Reference sites (Refs), two sites (WSF) were located on the Wollangambe River 0.2–1.2 km below the mine (Newnes States Forest) and two sites (WNP) were located 22 km below the mine (Wollemi National Park)

Order	Family	Refs.	WSF	WNP
Ephemeroptera	Baetidae	X		
	Coloburiscidae	X		X
	Leptophlebiidae	X		X
	Caenidae	X		
Plecoptera	Eustheniidae	X		X
	Gripopterygidae	X	X	X
	Austroperlidae	X		
Trichoptera	Hydrobiosidae	X		
	Glossomatidae	X		
	Hydroptilidae	X		
	Hydropsychidae	X	X	
	Ecnomidae	X		
	Conoesucidae	X		
	Calocidae	X		
	Leptoceridae	X		X
Oligochaeta	Oligochaeta	X	X	X
Hydracarina	Hydracarina	X	X	
Coleoptera	Hydaenidae	X		
	Elmidae (larvae)	X		X
	Elmidae (adult)	X		X
	Scirtidae	X		X
Diptera	Tipulidae	X	X	X
	Athericidae			X
	Ceratopogonidae	X		
	Simuliidae	X		X
	Empididae	X	X	X
	Chironomidae	X	X	X
Megaloptera	Corydalidae	X		X
Neuroptera	Neurorthidae	X		
Odonata	Aeshnidae	X		X
Lepidoptera	Pyralidae	X		
Collembola	Collembola		X	

community assemblages varied highly significantly according to the three site categories (Global R 0.558; $p < 0.001$). Pairwise comparison of site groups measured the extent of the differences in community assemblages downstream of the mine waste discharges. The samples collected immediately below the mine (WSF) had the most dissimilar invertebrate assemblages to reference sites (R-statistic 0.625, $p < 0.001$). Assemblage differences between references sites and sites further downstream (WNP) were less pronounced (R-statistic 0.308, $p < 0.001$). Comparison of assemblages below the mine discharge (WSF verses WNP) was strongly dissimilar (R-statistic 0.856, $p < 0.001$).

3.2 Water Quality

The coal mine waste water discharge was responsible for extensive changes to the chemistry of the Wollangambe River downstream of the mine. Electrical conductivity (EC) varied highly significantly ($p < 0.0001$; Table 2). Reference sites reported lowest EC (mean 25.6 μS/cm). The Wollangambe River at the two sites immediately below the discharge (WSF) mean EC was 386.5 μS/cm, 14 times higher than at the reference sites. Mean EC at the furthest downstream sites in the conservation area (WNP) was 246.3 μS/cm, almost ten times greater than the mean reference level (Table 2, Fig. 4).

The pH also varied highly significantly, according to site category ($p < 0.0001$; Table 2). The reference streams in this region are acidic (4.9–6.3), as it typical of many streams on Hawkesbury Sandstone geology (Tippler et al. 2012; Wright et al. 2015) (Table 2). This compares to the circum-neutral to alkaline levels (6.9–8.7) immediately downstream (WSF) of the mine waste discharge (as a consequence of the treatment process) and mildly acidic waters in the conservation area further downstream (WNP) (6.4–6.94; Table 2, Fig. 4).

Water temperature varied highly significantly according to sampling site category ($p < 0.0001$; Table 2). It was lowest at reference sites (mean 13.7 °C) and was highest (mean 16.5 °C) at the two sites immediately below the mine (WSF) (Table 2, Fig. 4). Dissolved oxygen levels also varied highly significantly according to sampling site category but the lowest levels (minimum 14.8%) were recorded at reference sites (mean 72.4%) and the highest levels were recorded at WNP 22 km below the mine (minimum 92.4%, mean 105%) (Table 2).

The ionic composition of the Wollangambe River was strongly modified downstream of the coal mine (Table 2). The concentration of all major ions varied significantly, according to sampling site category, with sodium the only exception (Table 2). The ionic composition of reference sites were sodium and chloride dominated (Table 2), with the cations (calcium, potassium

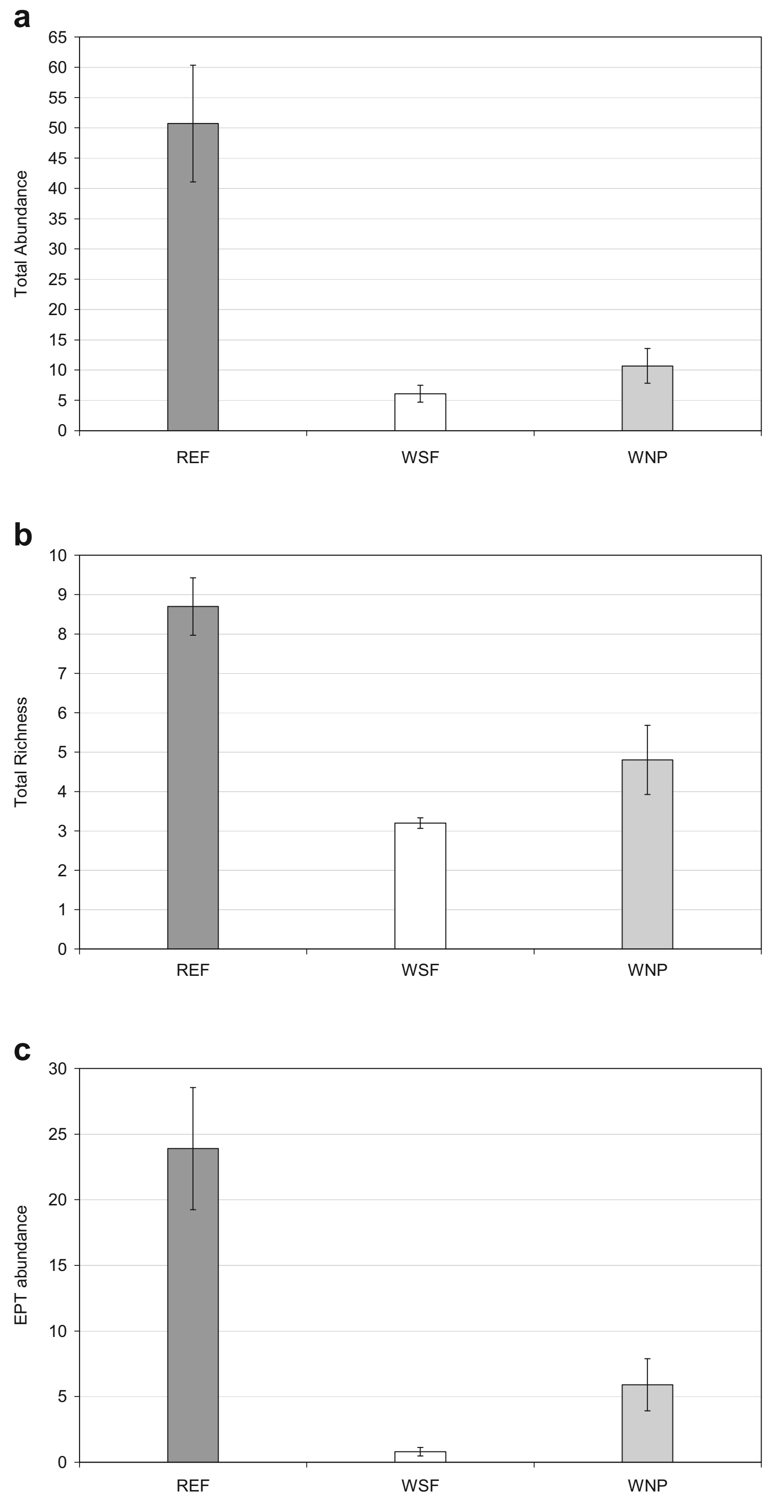


Fig. 2 Mean (plus/minus standard error) macroinvertebrate biotic indices. **a** *Top*, total abundance; **b** *middle*, total family richness; and **c** *bottom* EPT abundance. Results are provided for (*left*) reference sites (*REF*), *shaded dark grey*; (*centre*) Wollangambe River sampling sites in State Forest (0.2–1.2 km downstream mine waste) WSF, *unshaded*; and (*right*) Wollangambe River sampling sites in National Park (21–22 km downstream mine waste) WNP, *shaded light grey*

and magnesium) always below the detection limits and the anions sulphate and bicarbonate were only occasionally detected (Table 2). In comparison, locations on the Wollangambe River below the mine (WSF

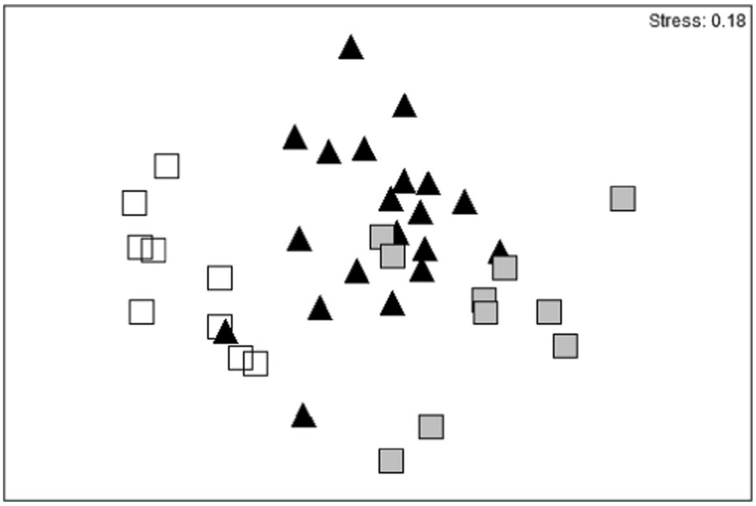


Fig. 3 *n*MDS ordination of macroinvertebrate samples. *Black triangles* represent reference samples. *Squares* represent samples collected from the Wollangambe River downstream of the mine waste discharge. *White squares* are samples from 0.2 to 1.2 km downstream (WSF) and *grey squares* from samples 21–22 km downstream (WNP)

and WNP) were calcium and sulphate dominated with sodium and chloride having lower mean concentrations than other major anions and cations (Table 2).

The concentration of the metals (zinc and nickel) varied highly significantly according to sampling site category ($p < 0.0001$; Table 2, Fig. 5). Reference sites reported the lowest mean concentration of zinc (4.0 μg/L) and nickel (0.7 μg/L). The highest concentrations of zinc and nickel in this study were found in the Wollangambe River below the mine discharge point, with mean zinc and nickel concentrations of 124.9 μg/L (Zn WSF) and 50.2 μg/L (Zn WNP) and 105 μg/L (Ni WSF) and 41.2 μg/L (Ni WNP) respectively (Table 2). The concentration of aluminium also varied significantly according to sampling site category ($p = 0.0001$; Table 2) but the trend was different to zinc and nickel. The concentration aluminium was highest at reference sites (mean 200.6 μg/L) and was lowest at WSF immediately below the coal mine (mean 26.7 μg/L; Table 2, Fig. 5).

4 Discussion

The Clarence Colliery's surface operation is located on land that is encircled by the GBMWHA. The location of the mine lies within 400 m of the boundary of the GBMWHA and the mine's waste water discharge point to the Wollangambe River is approximately 2 km upstream of the GMBWHA boundary. The entire mine lies within the upper catchment of the Wollangambe River and the steep topography ensures the rapid hydrologic connectivity of the mining operation to the protected lands and waterway (Pringle 2001). That is, all mine waste discharges or contaminated surface runoff only has to travel for a short time and distance before flowing into the protected World Heritage Conservation Area. Water quality and stream ecology data collected for this study confirmed the hydrologic connectivity and its impact for at least 22 km downstream from the mine, deep within the GBMWHA-protected area.

The disposal of the EPA licenced mine waste water from the Clarence Colliery directly into the Wollangambe River has a significant and negative environmental impact. The discharge of coal mine waste water causes substantial negative impacts to the aquatic ecology of the Wollangambe River. Macroinvertebrate communities were strongly degraded immediately below the mine discharge with some signs of ecological recovery 22 km further downstream. The current study found that taxonomic (family) richness dropped by more than 63% from a mean of 8.7 families per sample at reference sites to 3.2 families at the sites below (0.2–1.2 km) the mine. Such a reduction of family richness is larger than many coal mines regionally and internationally. For example, macroinvertebrate family richness dropped by only 20% downstream of the Tahmoor Colliery waste discharge, in the southern coal fields of the Sydney Basin (Wright et al. 2015). More similar to the current study, a reduction in taxonomic richness was observed in a study of West Virginia streams which reported approximately 50% less macroinvertebrate families at 'high' coal mining disturbance sites (Pond et al. 2008).

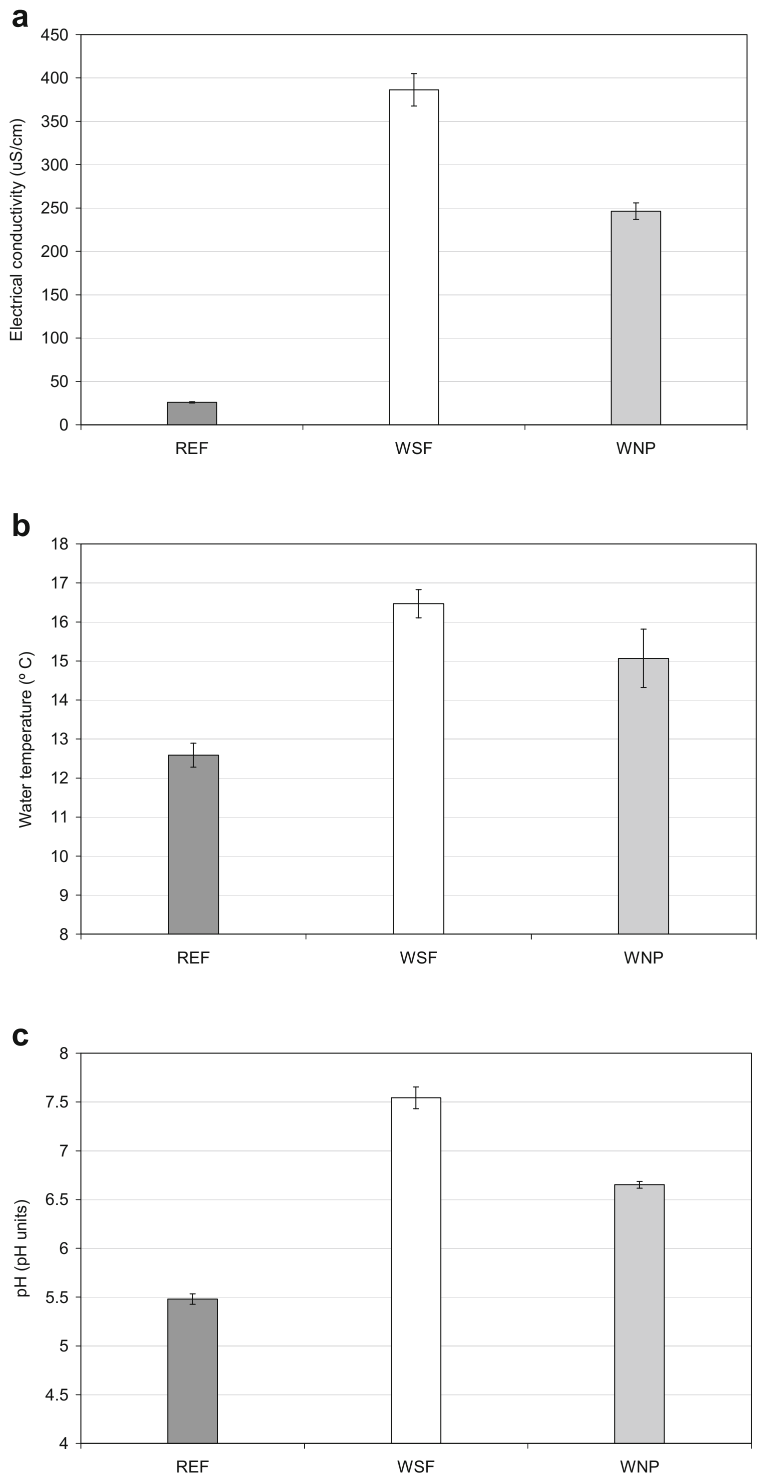
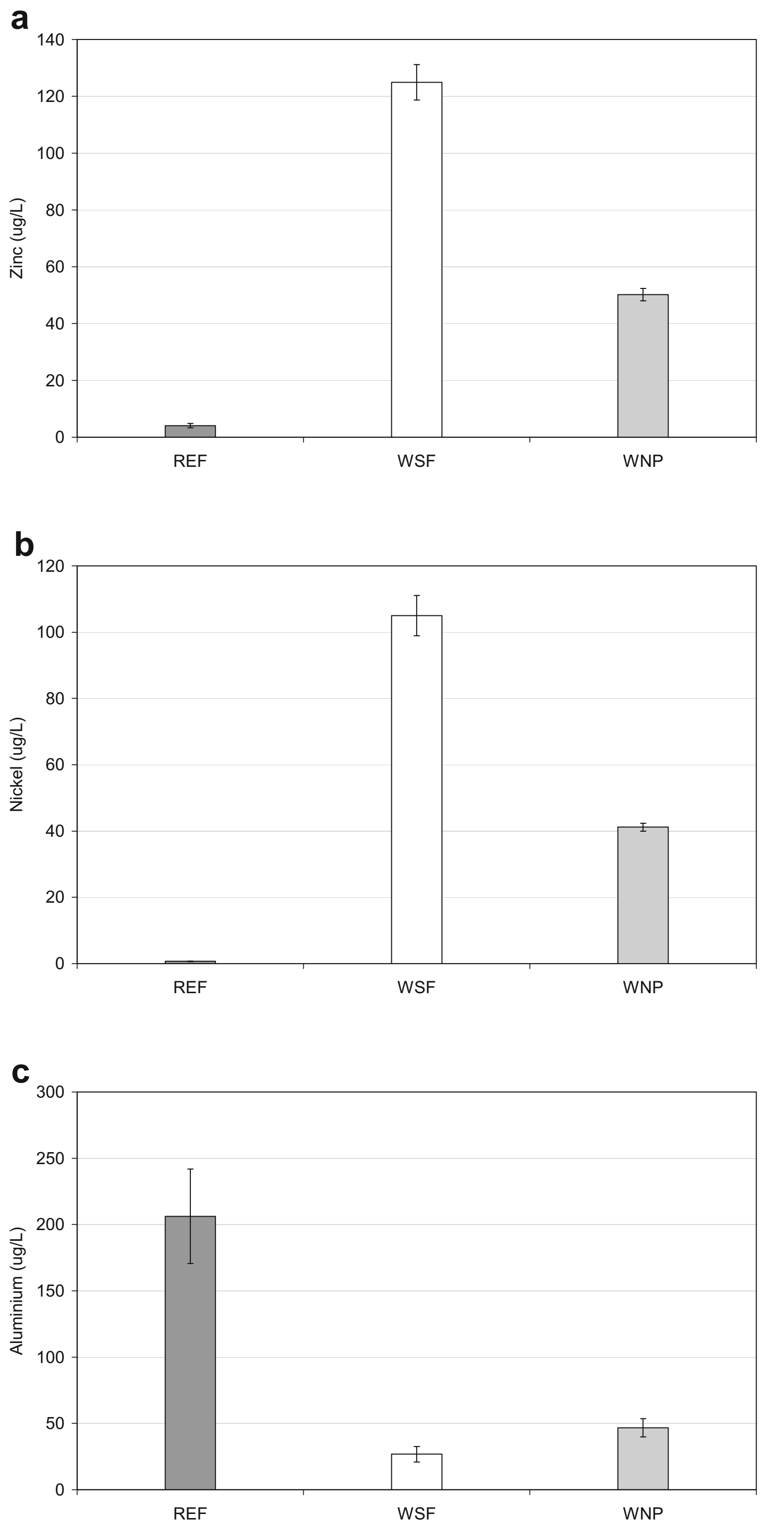


Fig. 4 Mean (plus/minus standard error) water quality. **a** *Top*, electrical conductivity; **b** *middle*, water temperature; and **c** *bottom*, pH. Results are provided for (*left*) reference sites (REF), *shaded dark grey*; (*centre*) Wollangambe River sampling sites in State Forest (0.2–1.2 km downstream mine waste) WSF, *unshaded*; and (*right*) Wollangambe River sampling sites in National Park (21–22 km downstream mine waste) WNP, *shaded light grey*

The current study recorded a very steep (96.7%) decline in the abundance of the pollution-sensitive EPT macroinvertebrates immediately below the coal mine. The average abundance of EPT macroinvertebrates at reference sites of 23.9 per sample fell to 0.8 per sample at the two sites immediately below the

Fig. 5 Mean (plus/minus standard error) metal concentrations in water samples for **a** *top*, total zinc; **b** *middle*, total nickel; and **c** *bottom*, total aluminium. Results are provided for (*left*) reference sites (REF), *shaded dark grey*; (*centre*) Wollangambe River sampling sites in State Forest (0.2–1.2 km downstream mine waste) WSF, *unshaded*; and (*right*) Wollangambe River sampling sites in National Park (21–22 km downstream mine waste) WNP, *shaded light grey*



discharge point. The longitudinal ecological impact remained evident with the EPT abundance increasing marginally to 5.9 per sample (75.3% less than reference sites), at the furthest sites (22 km) downstream of the mine discharge. This indicates that the ecological impairment in the Wollangambe River ecosystem below the coal mine is large in severity and geographic extent, with very few sensitive EPT animals able to survive below the mine. The reduction in EPT abundance at all sampling locations below the coal mine was considerably larger than was reported from a study of Rocky Mountain (CO, USA) streams polluted by heavy metals where EPT abundance in unpolluted reference streams (c. 190 invertebrates) dropped to c. 60 at the most highly metal polluted category (Clements et al. 2000). A comparatively smaller decline in EPT abundance was found in a study of West Virginia streams impacted by coal mines (Pond et al. 2008) with the proportion of macroinvertebrates in EPT groups at unmined streams of 77.9% compared to 51.1% at mined streams. The current study had a much greater negative impact on EPT macroinvertebrates than was observed in the nearby Bargo River where the proportion of EPT animals per sample dropped from 55% above the Tahmoor Colliery compared to 37.2% downstream (Price and Wright 2016).

An ecotoxicological study has confirmed that samples of Wollangambe River water, collected below the coal mine, were toxic to freshwater test species. In 2014, the NSW Office of Environment and Heritage conducted preliminary ecotoxicity testing of the Wollangambe River (from 1 km below the mine) and reported acute and chronic toxicity to the freshwater cladoceran *Ceriodaphnia dubia* at a range of dilutions. They also reported significant inhibitory effects were detected on growth of the freshwater green alga *Pseudokirchneriella subcapitata* (OEH 2015). Whilst these results indicate that ecotoxicity has been detected, further ecotoxicology testing is required to isolate the causative pollutants. Ecotoxicological methodologies for testing of coal mine wastes can help identify the individual pollutants responsible for the overall toxicity of the waste to a river ecosystem (e.g. Kennedy et al. 2005; Lattuada et al. 2009) and in turn can be used to set limits in a pollution licence.

The water chemistry modifications of the Wollangambe River due to the coal mine waste water was extensive. Of prominent ecological concern were elevated zinc and nickel concentrations in the Wollangambe River downstream of the mine. Both were much higher than reference samples collected upstream of the mine, or from unaffected tributaries. The background zinc concentration (at reference sites) was approximately 4 μg/L with nickel even lower (0.69 μg/L), often less than laboratory detection limits (<1 μg/L). The mean zinc concentration in the river directly below the mine waste outfall was 124 μg/L and nickel was 105 μg/L. The concentration of both metals fell with increased distance downstream, to mean concentrations of 50 μg/L (zinc) and 42 μg/L (nickel) at the lowest sites 22 km below the mine inflow. The concentrations of nickel and zinc that this study detected in the Wollangambe River below the mine discharge were at levels that are known to be hazardous for aquatic species, according to Australian water quality guidelines for protection of aquatic species (ANZECC 2000). Trigger values for freshwater streams to achieve 99 and 80% level of ecosystem protection, as reflected by per cent species affected, are ≤8 and ≤17 μg/L for nickel and ≤2.4 and ≤31 μg/L for zinc (ANZECC 2000). Water hardness will change the bioavailability of some metals, including both nickel and zine (ANZECC 2000). According the ANZECC guidelines (2000) the river immediately below the mine is classified as 'very hard' and 22 km downstream as 'moderately hard'. The ANZECC (2000) hardness corrected trigger value for 99% protection of species for nickel (ANZECC 2000) is ≤41.6 μg/L for very hard and ≤20 μg/L for moderate hardness and for zinc is ≤12.5 μg/L for very hard and ≤6 μg/L for moderate hardness. From this analysis and according to the ANZECC guidelines, both zinc and nickel are pollutants of ecological concern at all sampling sites downstream of the mine in the Wollangambe River. Only zinc is included in the EPL and permits 1500 μg/L (EPA 2016 EPL 729 at L2.4), well above the ANZECC guideline.

The concentrations of nickel and zinc in the Wollangambe River were intermediate when compared to other coal mine studies. Much higher levels of nickel and zinc were reported below the Green Valley coal mine (IN, USA) with nickel levels above 500 μg/L, and as high as 3780 μg/L and zinc often above 5000 μg/L (Brake et al. 2001). Higher levels of zinc and nickel from a disused coal mine (Canyon Colliery), about 10 km to the south-east of the Clarence coal mine, contaminated the receiving waterway (Dalpura Creek) with zinc (mean 305 μg/L) and nickel (mean 182.9 μg/L) (Price and Wright 2016). The current study showed that zinc and nickel concentrations were much higher than in coal mine-affected streams in West Virginia which had marginally elevated nickel in mined streams (mean 14.2 μg/L) compared to unmined streams (<10 μg/L) and almost identical zinc levels (<30 μg/L) in mined versus unmined streams (Pond et al. 2008). Locally, waste water from the Tahmoor mine caused lower zinc (mean 38.6 μg/L) and nickel (35.6 μg/L) enrichment of the Bargo River, according to samples collected 1 km downstream of the waste discharge (Wright et al. 2015).

The Wollangambe River has a naturally dilute ionic content with a consequent low salinity. The coal mine waste water increased EC of the river between 5 and 33 times downstream of the coal mine waste water discharge point (182 to 503 μS/cm) from background levels (15 to 36 μS/cm). This impact on salinity is consistent with the impacts reported by García-Criado et al. (1999) in north-western Spain with non-mining streams EC of 18 to 34 μS/cm compared to coal mine-affected streams of 202–472 μS/cm. Other studies have reported more significant increases in salinity from coal mine waste water. For example, Pond et al. (2008) reported for West Virginian coal mines an increase from 62 μS/cm in unmined streams to 1023 μS/cm downstream of mine waste water discharges. Coal mine discharges in the southern coal fields of the Sydney Basin cause larger increases in surface water salinity. For example, waste water discharge from Tahmoor Colliery increased the salinity of the Bargo River from 186 to 1078 μS/cm (Wright et al. 2015) and the West Cliff Colliery increased salinity in the Georges River from 173 μS/cm (upstream) to 1628 μS/cm below the mine waste discharge (Price and Wright 2016). Australian water quality guidelines provide a default guideline of <350 μS/cm for ecosystem protection of upland streams (south-eastern Australia) but recommends the calculation of site-specific guidelines (ANZECC 2000). Currently, the EPL for Clarence Colliery provides no discharge limits for salinity (EPL 726: EPA 2015).

The ionic composition of the Wollangambe River was strongly modified by the coal mine. Reference site water chemistry was dominated by sodium and chloride ions. This is similar to coastal flowing streams in south-eastern Australia which are dominated by sodium chloride (Hart and McKelvie 1986). Other streams in naturally vegetated catchments across the Sydney Basin have also reported the dominance of sodium and chloride ions (Wright 2012; Tippler et al. 2012). Above the mine, and at reference tributaries, sodium was the only cation detected and the anion dominance was $Cl > SO_4 > HCO_3$. Below the mine discharge the cation dominance was $Ca > Mg > Na = K$ and the anion dominance was $SO_4 > HCO_3 > Cl$. Below the mine, sulphate had a mean concentration of 178 mg/L, contrasting to reference sites where sulphate was often below the detection limit (<1 mg/L).

The sulphate concentrations below the mine is much higher than reported downstream of four other coal mines in the Sydney Basin, which had a mean sulphate concentrations of 20.4 mg/L (Wright 2012). However, the concentrations of sulphate in coal mine wastes in the Wollangambe River and other waterways affected by coal mines in the Sydney area is low when compared to international studies. For example, much higher sulphate levels have been reported in the USA below mountaintop mines in West Virginia (mean 695 mg/L) compared to higher background sulphate concentrations in unmined areas (mean 16 mg/L; Pond et al. 2008). UK coal mine waste waters have also been reported to contain very high sulphate concentrations of 404 to 1170 mg/L (Banks et al. 1997). Some of the highest sulphate concentrations reported have been in coal mines wastes in the Criciuma region of southern Brazil at levels as high as 8412 mg/L (Lattuada et al. 2009). Currently, the EPL for Clarence Colliery permits sulphate to be discharged in mine waste water at concentrations of up to 250 mg/L (EPL 726: EPA 2016). The comparatively low sulphate levels in the Wollangambe River, below the mine, is likely to be due to the generally lower content of sulphidic minerals in the coal ore. Australian coal, compared to other global coal ores, has a lower sulphur content (Huleatt 1991). Laboratory analysis of coal from the Katoomba Seam, which the Clarence Colliery extracts, has very low pyritic forms of sulphur of 0.02%, which is much lower than typically found in coal globally (Cohen 2002).

The discharge of mine waste water in this study caused a large increase in the pH of the Wollangambe River. The pH of the Wollangambe River (above the mine) and tributary reference sites are naturally acidic (mean 5.48) with a range of 4.9 to 6.3. Downstream of the mine discharge the pH increased by about 2 pH units to a mean of 7.54 (range of 6.9 to 8.7). This is a major point of difference between this study and others in the literature and differs to the widely observed reduction of pH due to coal mines drainage contaminated by AMD. For example, two streams containing coal mine effluent in the Hocking River (OH, USA) had mean pH levels of 3.0 and 3.1 (Verb and Vis 2000) with local unmined reference streams in the same catchment recording mean pH levels of 7.1 and 7.2 (Verb and Vis 2000). Similar results were reported from streams in the vicinity of the Green Valley mine (IN, 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). The cause of the increased pH appears to be due to Clarence Colliery waste water treatment. Mine water extracted from the underground workings at the Clarence Colliery has a reported pH range of 5.0–6.5 by Centennial Coal (1999) and 4.2 by Cohen (2002). Cohen (2002) explains that quick lime, calcium oxide, is added to one step of the waste water treatment process to increase water pH to very high levels to promote oxidation of the dissolved metals iron and manganese. The current mine EPL stipulates that waste water discharged from the mine must be in the pH range of 6.5–8.5 (EPL 726: EPA 2016). The planning consent for the mine has an approved water treatment process that adds calcium hydroxide to raise the pH of water to 9.2 and a further treatment system adds caustic soda to reduce the pH to within the licencing limits (Centennial Coal 2000).

The mine waste water discharges increased surface water temperature by 3.9 °C compared to reference sites. The increased temperature was most pronounced in the Wollangambe River immediately below the licenced discharge point. It is noted that the sampling was conducted in summer and autumn and did not include winter months that could amplify the thermal differences. This thermal impact was similar to the nearby inflow of waste seepage from the disused Canyon Coal mine which increased Dalpura Creek water temperature by 4.4 °C (Price and Wright 2016). Thermal water pollution is addressed in the Australian water quality guidelines as an ecological stressor (ANZECC 2000). The guidelines advocate site specific trigger values, using the 80th percentile of reference site water temperature. If this was applied to water temperature from the reference sites in this study, the 3.9 °C temperature increase would exceed this on all sampling occasions. Currently, the EPL for Clarence Colliery provides no discharge limits for waste water temperature (EPL 726: EPA 2016). However, thermal pollution of water has been regulated in NSW. For example, a large coal-fired power station, Vales Point Power Station, that is located about 120 km north-west of the study area discharges cooling water to Lake Macquarie, a large coastal estuary, does provide discharge conditions (EPL 761: EPA 2016) to limit thermal pollution in the receiving waters.

Clarence Colliery is located close to the boundary of, and discharges into, one of the most highly protected conservation areas and rivers in Australia. There are various statutory mechanisms at a federal and state government level that can provide protection to the Wollangambe River, the Wollemi National Park and more broadly the Greater Blue Mountains region from major activities such as coal mining. From a development approval and environmental pollution licencing perspective, the existing statutory mechanisms have failed to adequately protect the waterways and ecosystems, in spite of Clarence Colliery undergoing 18 licence variations and two clean up notices since 2001 (EPA 2016).

A window of pollution licence reform by the NSW Government has opened as a result of this research and other studies (Belmer et al. 2014; EPA 2015; NSW OEH 2015a). This study provides the additional evidence of the direct causation between the licenced waste water discharge from Clarence Colliery and its significant environmental impact up to (and probably well beyond) 22 km downstream from the mine waste outfall. It highlights the failure of environmental planning process and subsequent development assessment and pollution licencing controls in NSW, review and potential intervention by the Commonwealth with respect to the World Heritage obligations and the value of independent research that can be the catalyst for regulatory and industry reform. Within this protected region, the pollution from coal mines is not contained to the Clarence Colliery. Mine waste water from a nearby closed mine, Canyon Colliery, also located within the Blue Mountains World Heritage area, does not receive any treatment and has no pollution licencing requirements. The former Canyon mine has been found to be pollute the Grose River and is impacting on stream ecology (Wright and Burgin 2009a, b; Wright et al. 2011; Price and Wright 2016). This highlights the need for adequate pollution control, regulation and evaluation for current and former mines.

Case studies such as this are not without precedence. Other UNESCO-listed World Heritage areas damaged by mining include the 'Cradle of Humankind' World Heritage Site (Krugersdorp, South Africa), a sensitive karst system, that also contains one the world's most important paleontological and archaeological sites (UNESCO 2015). The integrity of that site is threatened by nearby gold and uranium mining that is associated with AMD and modifications to groundwater (Durand et al. 2010). Coal mining has been associated with degradation of conservation values of waterways not necessarily included in World Heritage estates. For example, water pollution from coal mining (and other activities such as coal-fired power stations, irrigation, light and heavy industries and urban settlements) in the catchment of the Olifants River in South Africa has been linked to the regional decline in the Nile crocodile (*Crocodylus niloticus*) (Ashton 2010). Water pollution from coal mining at the Matunuska Valley mines in Alaska in the late nineteenth century caused many wildlife impacts and destroyed culturally significant salmon streams (Weis 2014). These studies are, however, limited and represent an under-researched area.

This current study represents one of the most detailed assessments on the water quality and ecological responses to coal mine waste water discharges in an environmentally sensitive area containing internationally significant conservation values. In contrast to most other coal mine pollution studies, this case study traces the immediate and longitudinal water pollution impact of a single coal mine in a largely undisturbed catchment that is in an otherwise near-pristine condition. It highlights the failure of statutory process across two levels of government and lack of appropriate planning approvals and ongoing environmental pollution licencing.

References

ANZECC (Australian and New Zealand Environment and Conservation Council). (2000). *Australian and New Zealand guidelines for fresh and marine waters*. Canberra: Australian and New Zealand Environment and Conservation Council.

APHA (American Public Health Association). (1998). *Standard methods for the examination of water and wastewater* (20th ed.). Washington, DC: American Public Health Association.

Ashton, P. (2010). The demise of the Nile crocodile (*Crocodylus niloticus*) as a keystone species for aquatic ecosystem conservation in South Africa: the case of the Olifants. *Aquatic Conservation: Marine and Freshwater Ecosystems, 20*, 489–493.

Australian Commonwealth Government (2013). *Matters of national environmental significance. Significant impact guidelines 1.1 Environment Protection and Biodiversity Conservation Act 1999*. https://www.environment.gov.au/system/files/resources/42f84df4-720b-4dcf-b262-48679a3aba58/files/nes-guidelines_1.pdf. Accessed10 September 2016.

Australian Government (1998). The Greater Blue Mountains area: World Heritage nomination. Commonwealth Government of Australia and NSW National Parks and Wildlife Service. https://www.environment.gov.au/system/files/pages/50d276f9-337f-4d9f-85f5-120ded99fc85/files/gbm-nomination.pdf. Accessed10 September 2016.

Banks, D., Younger, P. L., Arnesen, R.-T., Iversen, E. R., & Banks, S. B. (1997). Mine-water chemistry: the good, the bad and the ugly. *Environmental Geology, 32*, 157–174.

Barbour, M. T., Plafkin, J. L., Bradley, B. P., Graves, C. G., & Wisseman, R. W. (1992). Evaluation of EPA's rapid bioassessment benthic metrics: metric redundancy and variability among reference stream sites. *Environmental Toxicology and Chemistry, 11*, 437–449.

Battaglia, M., Hose, G. C., Turak, E., & Warden, B. (2005). Depauperate macroinvertebrates in a mine affected stream: clean water may be the key to recovery. *Environmental Pollution, 138*, 132–141.

Birch, G., Siaka, M., & Owens, C. (2001). The source of anthropogenic heavy metals in fluvial sediments of a rural catchment: Coxs River, Australia. *Water, Air, and Soil Pollution, 126*, 13–35.

Belmer, N., Tippler, C., Davies, P. J., & Wright, I. A. (2014). Impact of a coal mine waste discharge on water quality and aquatic ecosystems in the Blue Mountains World Heritage Area. In G. Viets, I. D. Rutherfurd, & R. Hughes (Eds.), *Proceedings of the 7th australian stream management conference* (pp. 385–391). Townsville: the River Basin Management Society.

Brake, S. S., Connors, K. A., & Romberger, S. B. (2001). A river runs through it: impact of acid mine drainage on the geochemistry of West Little Sugar Creek pre- and post-reclamation at the Green Valley coal mine, Indiana, USA. *Environmental Geology, 40*, 1471–1481.

Centennial Coal (1999). *Clarence colliery proposed water management scheme*. Unpublished report, March. Prepared by International Environmental Consultants Pty. Ltd. Centennial Coal, Sydney.

Centennial Coal (2000). *Clarence colliery—lease extension. Environmental impact statement*. Unpublished reported, October. Prepared by International Environmental Consultants Pty. Ltd. Centennial Coal, Sydney.

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.

Clarke, K. R. (1993). Non-parametric multivariate analyses of changes in community structure. *Australian Journal of Ecology, 18*, 117–143.

Clements, W. H., Carlisle, D. M., Lazorchak, J. M., & Johnson, P. C. (2000). Heavy metals structure benthic communities in Colorado mountain streams. *Ecological Applications, 10*, 626–638.

Cohen, D. (2002). *Best practice mine water management at a coal mine operation in the Blue Mountains*. Masters of Engineering (Honours) thesis, University of Western Sydney, Penrith.

Corkum, L. D. (1989). Patterns of benthic invertebrate assemblages in rivers of north western North America. *Freshwater Biology, 21*, 191–205.

Cremin, A. (1989). The growth of an industrial valley: Lithgow, New South Wales. *Australian Historical Archaeology, 7*, 35–42.

Durand, J. F., Meeuvis, J., & Fourie, M. (2010). The threat of mine effluent to the UNESCO status of the Cradle of Humankind World Heritage Site. *The Journal for Transdisciplinary Research in Southern Africa, 6*, 73–92.

EPA (NSW Environment Protection Authority) (2015). Variation to an environment protection licence for Clarence Colliery. http://www.epa.nsw.gov.au/licensing/clarence-colliery.htm. Accessed 17 January 2017.

EPA (NSW Environment Protection Authority). (2016). Search for environment protection licences. http://www.epa.nsw.gov.au/prpoeoapp/default.aspx. Accessed 10 September 2016.

Fairweather, P. G. (1990). Sewage and the biota on seashores: assessment of impact in relation to natural variability. *Environmental Monitoring and Assessment, 14*, 197–210.

García-Criado, F., Tomé, A., Vega, F. J., & Antolín, C. (1999). Performance of some diversity and biotic indices in rivers affected by coal mining in northwestern Spain. *Hydrobiologia, 394*, 209–217.

Graham, K., & Wright, I. A. (2012). The potential and reality of the environment protection licensing system in NSW: the case of water pollution. *Environmental Planning and Law Journal, 29*, 359–372.

Gray, D. P., & Harding, J. S. (2012). Acid Mine Drainage Index (AMDI): a benthic invertebrate biotic index for assessing coal mining impacts in New Zealand streams. *New Zealand Journal of Marine and Freshwater Research, 46*, 335–352.

Griffith, M. B., Norton, S. B., Alexander, L. C., Pollard, A. I., & LeDuc, S. D. (2012). The effects of mountaintop mines and valley fills on the physicochemical quality of stream ecosystems in the central Appalachians: a review. *Science of the Total Environment, 417–418*, 1–12.

Harrison, J., Heijnis, H., & Caprarelli, G. (2003). Historical pollution variability from abandoned mine sites, Greater Blue Mountains World Heritage Area, New South Wales, Australia. *Environmental Geology, 43*, 680–687.

Hart, B. T., & McKelvie, I. D. (1986). Chemical limnology in Australia. In W. D. Williams & P. P. De Deckker (Eds.), *Limnology in Australia* (pp. 3–31). Collingwood: CSIRO Publishing.

Hawking, J. H. (2000). *A preliminary guide to keys and zoological information to identify invertebrates from Australian freshwaters. Identification Guide No. 2* (2nd ed.). Albury: Cooperative Research Centre for Freshwater Ecology.

Hickey, C. W., & Clements, W. H. (1998). Effects of heavy metals on benthic macroinvertebrate communities in New Zealand streams. *Environmental Toxicology and Chemistry, 17*, 2338–2346.

Huleatt, M. B. (1991). *Handbook of australian black coals: geology, resources, seam properties and product specifications*. Bureau of mineral resources, geology and geophysics. Canberra: Australian Government Publishing Service.

Jarvis, A. P., & Younger, P. L. (1997). Dominating chemical factors in mine water induced impoverishment of the invertebrate fauna of two streams in the Durhan Coalfield, UK. *Chemistry and Ecology, 13*, 249–270.

Johnson, D. B. (2003). Chemical and microbiological characteristics of mineral spoils and drainage waters at abandoned coal and metal mines. *Water, Air, and Soil Pollution, 3*, 47–66.

Johnson, D. B., & Hallberg, K. B. (2005). Acid mine drainage remediation options: a review. *Science of the Total Environment, 338*, 3–14.

Keith, D. A., & Benson, D. H. (1988). Natural vegetation of the Katoomba area. *Cunninghamia, 2*, 107–144.

Kennedy, A. J., Cherry, D. S., & Zipper, C. E. (2005). Evaluation of ionic contribution to the toxicity of a coal-mine effluent using *Cerodaphnia dubia. Archives of Environmental Contamination and Toxicology, 49*, 155–162.

Lattuada, R. M., Menezes, C. T. B., Pavei, P. T., Peralba, M. C. R., & Dos Santos, J. H. Z. (2009). Determination of metals by total reflection X-ray fluorescence and evaluation of toxicity of a river impacted by coal mining in the south of Brazil. *Journal of Hazardous Materials, 163*, 531–537.

Lenat, D. R. (1988). Water quality assessment of streams using a qualitative collection method for benthic macroinvertebrates. *Bulletin of the North American Benthological Society, 7*, 222–233.

Lenat, D. R., & Penrose, D. L. (1996). History of the EPT taxa richness metric. *Bulletin of the North American Benthological Society, 13*, 305–307.

Macqueen, A. (1997) *Back from the brink: Blue gum forest and the Grose wilderness*. Self-published. Bee Farm Road, Springwood.

Merriam, E. R., Petty, J. T., Merovich, G. T., Jr., Fulton, J. B., & Strager, M. P. (2011). Additive effects of mining and residential development on stream conditions in a central Appalachian watershed. *Journal of the North American Benthological Society, 30*, 399–418.

Metzeling, L., Perriss, S., & Robinson, D. (2006). Can the detection of salinity and habitat simplification gradients using rapid bioassessment of benthic invertebrates be improved through finer taxonomic resolution or alternatives indices? *Hydrobiologia, 572*, 235–252.

NSW OEH (NSW Office of Environment and Heritage) (2015a). *Canyon Colliery discharge investigation*. Office of Environment and Heritage: 59 Goulburn Street Sydney NSW 2000. http://www.epa.nsw.gov.au/resources/licensing/150171-clarence-colliery-discharge-investigation.pdf. Accessed 17 January 2017.

NSW OEH) (NSW Office of Environment and Heritage) (2015b). *Wild rivers*. http://www.environment.nsw.gov.au/parktypes/wildrivers.htm. Accessed 9 September 2016.

Plafkin, J. L., Barbour, M. T., Porter, K. D., Grosse, S. K., & Hughes, R. M. (1989). *Rapid bioassessment protocols for use in streams and rivers: benthic macroinvertebrates and fish*. Washington DC: United States Environmental Protection Agency.

Pond, G. J., Passmore, M. E., Borsuk, F. A., Reynolds, L., & Rose, C. (2008). Downstream effects of mountaintop coal mining: comparing biological conditions using family- and Genus-level macroinvertebrate bioassessment tools. *Journal of the North American Benthological Society, 27*, 717–737.

Price, P., & Wright, I. A. (2016). Water quality impact from the discharge of coal mine wastes to receiving streams: comparison of impacts from an active mine with a closed mine. *Water, Air, and Soil Pollution*. doi:10.1007/s11270-016-2854-7.

Pringle, C. M. (2001). Hydrologic connectivity and the management of biological reserves: a global perspective. *Ecological Applications, 11*, 981–998.

Resh, V. H., & Jackson, J. K. (1993). Rapid assessment approaches to biomonitoring using benthic macroinvertebrates. In D. M. Winter & V. H. Resh (Eds.), *Freshwater biomonitoring and benthic macroinvertebrates* (pp. 195–223). Chapman & Hall: New York.

Rosenberg, D. M., & Resh, V. H. (1993). *Freshwater biomonitoring and benthic macroinvertebrates*. New York: Chapman & Hall.

Scullion, J., & Edwards, R. W. (1980). The effects of coal industry pollutants on the macroinvertebrate fauna of a small river in the South Wales coalfield. *Freshwater Biology, 10*, 141–162.

Tippler, C., Wright, I. A., & Hanlon, A. (2012). Is catchment imperviousness a keystone factor degrading urban waterways? A case study from a partly urbanised catchment (Georges River, south-eastern Australia). *Water, Air, and Soil Pollution, 223*, 5331–5344.

Underwood, A. J. (1991). Beyond BACI: experimental designs for detecting human environmental impacts on temporal variations in natural populations. *Australian Journal of Marine and Freshwater Research, 42*, 569–587.

UNESCO (2009). *Greater Blue Mountains area*. United Nations Educational, Scientific, and Cultural Organisation. http://whc.unesco.org/en/list/917/. Accessed 10 September 2016.

UNESCO (2015). *Fossil hominid sites of South Africa*, United Nations Educational, Scientific, and Cultural Organisation. http://whc.unesco.org/en/list/915. Accessed 10 September 2016.

Verb, R. G., & Vis, M. L. (2000). Comparison of benthic diatom assemblages from streams draining abandoned and reclaimed coal mines and non-impacted sites. *Journal of the North American Benthological Society, 19*, 274–288.

Warwick, R. M. (1993). Environmental impact studies on marine communities: pragmatical considerations. *Australian Journal of Ecology, 18*, 63–80.

Weis, S. (2014). The local and world-wide impact of mining Alaska's coal. *Environment: Science and Policy for Sustainable Development, 56*, 25–29.

NP&WS (NSW National Parks and Wildlife) (2001a). *Blue Mountains National Park plan of management*. http://www.environment.nsw.gov.au/resources/parks/pomfinalbluemountains.pdf. Accessed 9 September 2016.

NP&WS (NSW National Parks and Wildlife) (2001b). *Wollemi national park plan of management*. http://www.environment.nsw.gov.au/resources/parks/pomFinalWollemi.pdf. Accessed 9 September 2016.

Winterbourn, M. J. (1998). Insect faunas of acidic coal mine drainages in Westland, New Zealand. *New Zealand Journal of Entomology, 21*, 65–72.

Wright, I. A. (2012). Coal mine 'dewatering' of saline wastewater into NSW streams and rivers: a growing headache for water pollution regulators. In J.R. Grove, & I.D. Rutherfurd (Eds). *Proceedings of the 6th Australian Stream Management Conference: Managing for Extremes*, 6–8 *February, 2012, Canberra, Australia*, (pp 206–213). Published by the River Basin Management Society.

Wright, I. A., & Burgin, S. (2009a). Comparison of sewage and coal-mine wastes on stream macroinvertebrates within an otherwise clean upland catchment, south-eastern Australia. *Water, Air, and Soil Pollution, 204*, 227–241.

Wright, I. A., & Burgin, S. (2009b). Effects of organic and heavy-metal pollution on chironomids within a pristine upland catchment. *Hydrobiologia, 635*, 15–25.

Wright, I. A., & Ryan, M. (2016). Impact of mining and industrial pollution on stream macroinvertebrates: importance of taxonomic resolution, water geochemistry and EPT indices for impact detection. *Hydrobiologia, 772*, 103–115.

Wright, I. A., Chessman, B. C., Fairweather, P. G., & Benson, L. J. (1995). Measuring the impact of sewage effluent on the macroinvertebrate community of an upland stream: the effect of different levels of taxonomic resolution and quantification. *Australian Journal of Ecology, 20*, 142–149.

Wright, I.A., Wright S.A., Graham, K., & Burgin, S. (2011). Environmental protection and management: a water pollution case study within the Greater Blue Mountains World Heritage Area. *Land Use Policy, 28*, 353–360.

Wright, I. A., McCarthy, B., Belmer, N., & Price, P. (2015). Subsidence from an underground coal mine and mine wastewater discharge causing water pollution and degradation of aquatic ecosystems. *Water, Air, & Soil Pollution, 226*, 1–14.

Younger, P. L. (2004). Environmental impacts of coal mining and associated wastes: a geochemical perspective. *Geological Society, 236*, 169–209.