



# Legacy Contamination of River Sediments from Four Decades of Coal Mine Effluent Inhibits Ecological Recovery of a Polluted World Heritage Area River

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**Abstract** A revised environmental licence that authorises the disposal of coal mine effluent has reduced the severity and spatial extent of water pollution and associated ecological impairment of a high conservation-value river flowing into and within the Greater Blue Mountains World Heritage Area. This study investigated water quality and the ecological condition of the Wollangambe River above and below a colliery effluent outfall and assessed the longitudinal impact 22 km downstream. Results are compared to a previous study conducted in 2012/2013 when the environmental licence allowed hazardous pollutant discharges (zinc, nickel) from the colliery. The current study revealed that water quality and river sediment at sampling sites in close proximity (< 2 km) to the effluent outfall continues to contribute ecologically hazardous concentrations of metals and river macroinvertebrates reflect diminished ecological

health. However, further downstream monitoring has revealed a significant improvement in ecological condition that can be directly attributed to the revised pollution licence. We hypothesise that the ecological recovery of the most contaminated reaches of the river that lies proximate to the discharge point is constrained by four decades of accumulated zinc and nickel within river sediments. Nickel (978 mg/kg) and zinc (2080 mg/kg) exceeded ecosystem protection guidelines by 45 and 10 times, respectively. The study highlights the importance of appropriate and site-specific environmental licencing to protect riverine ecosystems of conservation significance from long-term contamination.

**Keywords** World heritage · Environmental management · Freshwater ecosystems · Water quality · Macroinvertebrates · Metal pollution · Ecological recovery · Coal mine wastewater · Wastewater regulation

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## 1 Introduction

Underground coal mining has been an important industry in the Blue Mountains region of NSW, Australia, for more than a century (Cremin, 1989; Macqueen, 1997). This region currently has two active and several closed coal mines that impair water quality and biota of local streams and rivers (Battaglia et al., 2005; Wright & Burgin, 2009a, b; Wright et al.,

2017; Belmer & Wright, 2020). Contaminated water from coal mining operations is a well-known cause of water pollution internationally, including the USA (Skousen et al., 2019), the UK (Younger, 2004), China (Wei et al., 2013) and New Zealand (Gray & Harding, 2012). Coal mine effluent can be categorised as *acid mine drainage* (AMD) where it has low (< 5) pH (Younger, 2004). Coal effluent often has elevated salinity (Daniels et al., 2016; Belmer & Wright, 2020) and elevated concentrations of metals and metalloids (Strosnider et al., 2020). Coal mine operations are widely associated with both impairment of water quality and adverse impacts to river biota (Hickey & Clements, 1998; Gray & Harding, 2012).

A series of water quality and ecological impacts have been reported in rivers and streams affected by active and closed coal mines in the Blue Mountains region of the Sydney Basin's western coalfields (Birch et al., 2001; Harrison et al., 2003; Battaglia et al., 2005; Wright & Burgin, 2009a, b; Belmer et al., 2014; Price & Wright, 2016; Wright et al., 2017; Belmer & Wright, 2020). For example, coal mine drainage from untreated wastes from closed coal mines has caused low pH (5.1) and elevated salinity, zinc and nickel concentration within Neubecks Creek, Lithgow (Birch et al., 2001; Battaglia et al., 2005). Neubecks Creek also had impaired ecological health, based on lower abundance and taxonomic richness of stream macroinvertebrates (Battaglia et al., 2005). Pollution from coal mine effluent has also impacted high conservation-value waterways within the Greater Blue Mountains World Heritage Area (GBMWA) (Wright & Burgin, 2009a, b; Belmer et al., 2014; Price & Wright, 2016). These studies documented elevated zinc and nickel concentrations at ecologically hazardous concentrations (nickel 210 µg/l; zinc 370 µg/l) (Price & Wright, 2016) flowing freely and without any treatment, from the drainage adit of the closed Canyon Colliery mine into Dalpura Creek, within the GBMWA protected area. The concentrations of zinc and nickel were measured at ecologically hazardous levels for more than 6 km downstream and are associated with the ecological impairment of the 'protected' Grose River (Wright & Burgin, 2009a, b). The adverse impact to the Grose River's ecological health was also assessed through reduced invertebrate taxonomic richness, abundance and reduction of pollution-sensitive groups, such as Ephemeroptera, Plecoptera and Trichoptera (EPT) families (Lenat

& Penrose, 1996) and modified chironomid species assemblages (Wright and Burgin (2009a, b). The pollution of the Grose River from the closed Canyon Colliery was enabled through a number of weaknesses in the environmental and planning regimes that should have regulated water pollution from both active and closed coal mines (Wright et al., 2011).

The underground Clarence Colliery has disposed of its effluent into the headwaters of the Wollangambe River since it began operation in 1980 (Wright et al., 2017). In 2020, the mean monthly discharge from October to December ranged from 10.7 to 15.6 ML/day (Centennial Coal 2021). It is likely that similar volumes of wastewater discharged from this colliery are typical, with an earlier study by Cohen (2002) estimating that 14 ML a day was routinely released. The mine effluent receives limited dilution from the upstream Wollangambe River (NSW OEH 2015). Mine water sampled from *goaf* (previously mined) areas within underground workings of the Clarence Colliery, along with the nearby (now closed) Canyon Colliery, was previously investigated by Judell and Anderson (1988). Both collieries mined the Katoomba seam of the Illawarra coal measures. The study confirmed that the untreated mine drainage waters conform to *acid mine drainage* due to low pH (3.0–4.2) and elevated sulphate (32–270 mg/L; Judell & Anderson, 1988). In addition, the *goaf* waters from these mines also had highly elevated zinc concentrations (1500–8300 µg/l) (Judell & Anderson, 1988). The Clarence mine drainage is treated before it is disposed to the Wollangambe River, with Cohen (2002) describing an earlier multi-stage treatment process that was used at the colliery. The treatment of Clarence Colliery effluent has been upgraded to increase pH and reduce concentration of several pollutants, such as zinc and nickel, in accordance with its Environment Protection Authority (EPA) licence (EPL 726: EPA 2021). During the 2020 study, the pH of mine effluent ranged from 8.0 to 8.5 (Clarence Colliery 2021). Consequently, although the mine generates *acid mine drainage*, its treated effluent disposed into the Wollangambe River is alkaline.

The contamination triggered by the waste discharge impairs an otherwise largely undisturbed river and an area of very high conservation land that managed by several statutory mechanisms, at both the state and national level, to protect its natural environmental values as a national park, declared wild

river, wilderness area and part of the GBMWA (Australian Government, 1998). An earlier investigation of Wollangambe River found Clarence Colliery wastes increased concentrations of nickel (mean 105 µg/l) and zinc (mean 124.9 µg/l) in the Wollangambe River downstream of the mine outfall (Belmer et al., 2014; Wright et al., 2017). The concentration of these metals exceeded the Australian ecological protection guidelines (ANZECC (Australian and New Zealand Environment and Conservation Council), 2000) for at least 22 km downstream of the outfall. At reference sites, the background concentration of both metals was at much lower concentrations (nickel < 1 µg/l; zinc < 5 µg/l) (Belmer et al., 2014; Wright et al., 2017). Additionally, the mine effluent discharge caused severe impairment of the Wollangambe River with a decrease in river invertebrate taxonomic (family) richness by more than 63% at the sites immediately below the mine outfall, compared to unaffected reference sites (Wright et al., 2017). An earlier study by Cohen (2002) conducted a detailed investigation of Wollangambe River sediment, above and below the mine outfall. That study revealed that elevated concentrations of metals (cobalt, iron, manganese, nickel and zinc) were detected at potentially ecologically hazardous concentrations downstream of the effluent outfall (Cohen, 2002).

Pollution of the Wollangambe River from wastes disposed from the Clarence Colliery was internationally recognised as being of heightened concern in 2020, due to the nature and extent of the pollution impact and also due to the high threat to the GBMWA outstanding universal conservation values (IUCN, 2020). This was despite assurances within the Australian government's 1998 nomination document for World Heritage listing of the area that explained how coal mines in the Blue Mountains area were subject to strict environmental regulation and would not affect GBMWA catchments (Australian Government, 1998). In December 2020, water pollution from coal mines including both Clarence Colliery and the closed Canyon Colliery was identified as a *high threat* that contributed to the downgrade in the conservation outlook of GBMWA, from *good with some concerns* to *significant concern* (IUCN, 2020). This was in addition to the largest *very high threat* to the conservation outlook for GBMWA from frequent high-intensity bushfires, particularly after a series of severe fires

over summer 2019/2020 that burnt about 82% of the area (IUCN, 2020). UNESCO recognises that the cumulative impact of future mining projects in the surrounding region, in addition to existing mines, may threaten the world heritage values of the GBMWA (UNESCO, 2021a).

Disposal of coal mine effluent to the Wollangambe River, and other aspects of the environmental performance of the colliery operation, is regulated by the NSW Environment Protection Authority (NSW EPA). The regulation and control of water pollution from waste discharges in NSW from coal mines, and from other industrial operations, use a pollution licencing approach. This was based on a command-and-control approach and has shifted to a risk-based framework (EPA 2013). Site-specific environment protection licences (EPLs) form the foundation to environmental regulation of specific scheduled industries (as defined by the *Protection of the Environment Operations Act 1997* (NSW), such as coal mines, which specifies the allowable pollutants, pollutant concentration and permitted volume for effluent point source discharges to the environment (Graham & Wright, 2012; EPA, 2013). The NSW EPA regulation of industrial effluent discharges is similar to the permit approach used to administer wastewater discharges in the USA under the US Clean Water Act (1972; Rich 2016). The specific pollutant concentrations for the Clarence Colliery discharge are contained in EPL 726 (EPA, 2021).

For nearly 25 years of operation, the NSW EPA did not apply any restrictions to hazardous pollutants, such as zinc or nickel, to effluent discharge limits from the colliery to the Wollangambe River (EPL 726; EPA, 2021). From 2004 to 2020, the NSW EPA conducted numerous revisions of the colliery's pollution licence and progressively reduced the concentration of zinc and then nickel, along with 20 other pollutants. In 2004, zinc was first included in the NSW EPA regulations for the colliery effluent with a maximum allowable concentration of 5000 µg/l (EPL 726; EPA, 2021; Notice # 1037161). In subsequent variations, from 2004 to 2020, the permitted concentration of zinc in the effluent was reduced, firstly to 2500 µg/l in 2011 (EPL 726; EPA, 2021; Notice # 1502867), then 1500 µg/l in 2014 (EPL 726: EPA, 2021; Notice # 1521882) and finally to 8 µg/l in January 2020 (EPL 726; EPA, 2021; Notice # 1590599). Nickel was not included in the NSW EPA regulations

for Clarence Colliery until January 2020 when it was first included at a maximum concentration of 11 µg/l.

This current study investigated water pollution and ecological health of the Wollangambe River in 2020 and compared to results to those from an earlier 2012/2013 study. 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 methods followed those used in an earlier investigation in 2012/2013 which was conducted 7.5 years previously (Belmer et al., 2014; Wright et al., 2017). The current study was framed around answering four questions:

1. Has water quality and stream macroinvertebrate assemblages of the Wollangambe River changed since the previous investigation in 2012/2013?
2. Can any of the water quality and stream macroinvertebrate assemblage changes in (1) be attributed to the progressive 2004–2020 revision of the Clarence Colliery EPL regulating the pollutant concentrations in the coal mine wastewater discharges?
3. What are the concentrations of metals in Wollangambe River sediment and does this influence water quality and ecological health of the river?
4. Does any section of the Wollangambe River show any signs of water quality or ecological health ‘recovery’ from previously documented pollution effects (from 2012/2013 investigation), particularly within the GBMWhA?

## 2 Methods

### 2.1 Study Area

The Clarence Colliery is located within the western coalfields of the Sydney Basin (Mudd, 2009). The entire mining operation lies within the upper hydrological catchment of the Wollangambe River, and the steep topography ensures the rapid hydrologic connectivity of runoff and liquid wastes from the mining operation to the protected lands and waterways (Pringle, 2001). Mine effluent discharges, once released, travel approximately 200 m via an unnamed tributary to the Wollangambe River. The confluence where the mine effluent enters the river is a short

distance (c. 1.6 km) upstream of the World Heritage boundary. Water quality, bioaccumulation and stream ecology results from previous studies (Belmer et al., 2014, 2019; Wright et al., 2017) confirm that the effluent discharge has caused substantial environmental harm to at least 22 km of the Wollangambe River, within the ‘protected’ GBMWhA area.

### 2.2 Study Design and Sampling Sites

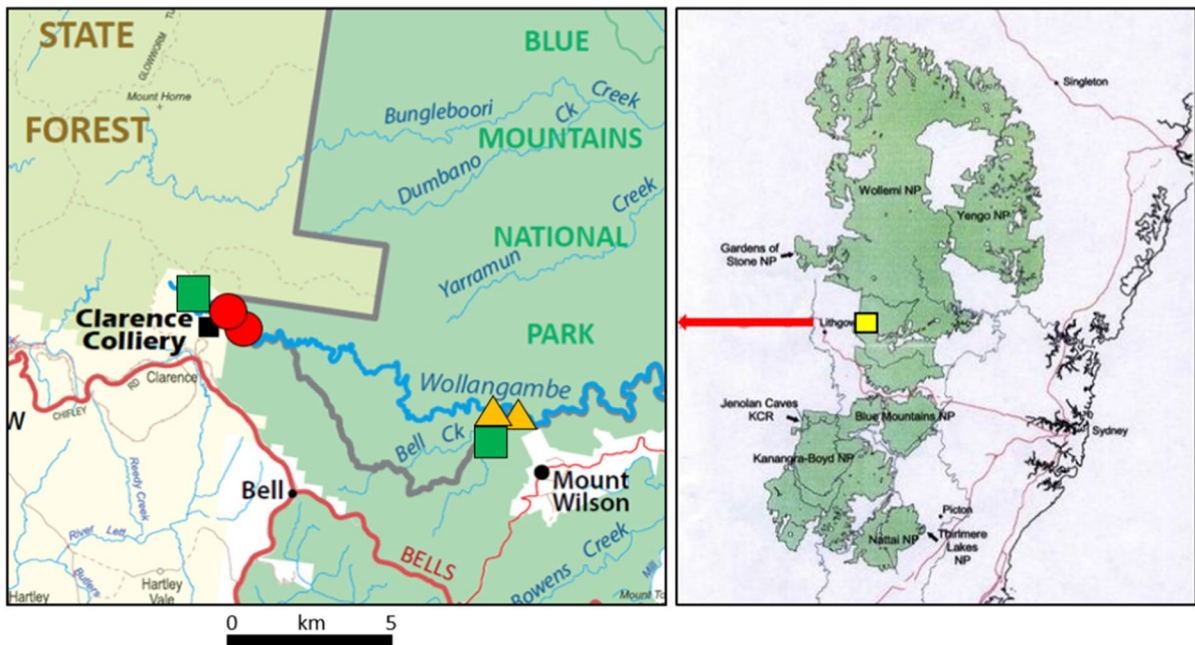
River sediment samples were also collected and tested for total metal content, following the sediment study by Cohen (2002). Water quality, river sediment and stream invertebrate samples were collected from two reference sites to enable comparison with physically similar sites (size, altitude, geology and stream habitats) exposed to the coal mine wastewater (Fairweather, 1990). The previous water quality and stream invertebrate study had collected samples from a total of four reference sites (Wright et al., 2017). However, an inspection of two sites resulted in their exclusion as it was revealed that a combination of drought and bushfire over the previous few years (Godfree et al., 2021) had caused sedimentation of the creek channels and had reduced surface flow.

Sampling was carried out in October, November and December 2020 on three upland streams or rivers (Table 1) across six sampling locations within the upper Wollangambe River catchment (11,138 ha) in the Greater Blue Mountains area (33°28’S, 150°17’E) (Figure 1). Four sites were located along the Wollangambe River. Sampling sites were grouped into pairs of sites. The first pair were situated downstream of the mine effluent, one on an unnamed tributary that carried the effluent and the other in the Wollangambe River approximately 1 km downstream of the effluent outfall. Both were in close proximity (< 2 km) of the mine waste outfall. This sampling pair is referred to as *Wollangambe State Forest* (WSF) being located within NSW State Forest land immediately upstream of the GBMWhA boundary (Table 1). The second pair was located c. 22 km below the mine waste outfall. This pair of sites is referred to as *World Heritage Area* (WhA) (Table 1). The sampling design included two ‘reference’ sites that were unaffected by coal mine effluent. The two reference sites included one site upstream of the mine discharge at the headwaters of Wollangambe River (W1) and the other on



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

Site category, location and name	Coordinates	Width	Stream order	Altitude (m ASL)
REF. Wollangambe River 200 m above entry of colliery effluent	33° 27'24.3"S, 150° 14'58.19"E	0.5–1 m	1st	992
WSF. Unnamed tributary 250 m below effluent outfall	33° 27'28.8"S, 150° 14'58.78"E	1–2 m	1st	980
WSF. Wollangambe River 1.2 km below effluent outfall	33° 27'20.37"S, 150° 15'27.43"E	1.5–2 m	2nd	956
WHA. Wollangambe River 21.5 km below effluent outfall	33° 29'21.07"S, 150° 21'13.74"E	2–3 m	2nd	745
WHA. Wollangambe River 22 km below coal mine discharge.	33° 29'15.18"S, 150° 21'20.98"E	4–5 m	3rd	741
REF Bell Creek	33° 29'23.97"S, 150° 21'14.84"E	1–2 m	2nd	748



**Figure 1** Map showing the study area (at left) and location within (yellow square) the Greater Blue Mountains World Heritage Area (at right). Reference sites are green squares. WSF

sites (< 2 km below Clarence Colliery effluent outfall) are red circles. WHA sites (22 km below effluent outfall) are gold triangles

Bell Creek, a tributary of the Wollangambe River near the WHA impact sites (Figure 1, Table 1).

The 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) (Figure 1, Table 1). Bell Creek (Figure 1), in particular, drained a similar sub-catchment to the upper Wollangambe River and was considered to have similar size and flow characteristics to the Wollangambe River.

Aquatic macroinvertebrates were collected to measure any freshwater ecological health changes associated with the coal mine waste discharges. Macroinvertebrate samples were collected from five of the six sites using identical methods to those previously used in the 2012/2013 study (Belmer et al., 2014; Wright et al., 2017). The unnamed waterway that conveyed the mine effluent (WSF) was only sampled for water and sediment, not macroinvertebrates (Table 1). For a detailed description of the macroinvertebrate sampling and laboratory methods, please see Wright

et al. (2017) as identical methods were used in the current study.

Water chemistry was investigated on three occasions in October, November and December 2020. On each occasion, 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. Duplicate water “grab” samples were collected in clean and unused sample containers provided by a National Association of Testing Authorities (NATA) accredited laboratory.

At each sampling site, three or four random samples of river sediment were collected at each site in a zone of accumulated sediment. This resulted in six to eight samples for each sampling category. The sampling methods used followed those recommended by the Victoria EPA (Victoria EPA 2009). Samples were placed into sealed glass sampling jars and stored for subsequent chemical analysis of metal content by the Envirolab laboratory. As for water samples, the sediment samples were placed for storage in a cooled and insulated container. Samples were chilled and delivered to the laboratory for analysis.

All water samples and sediment samples were analysed using standard methods (APHA, 1998) by the NATA accredited laboratory for major anions, major cations and total metals (zinc, nickel, aluminium and 12 other metals). Analytical QA/QC procedures within the testing laboratory included the use of sample blanks and spiked samples to ensure the reliability of analytical procedures.

### 2.3 Biotic Indices and Data Analysis

Macroinvertebrate family richness, total abundance and EPT (Ephemeroptera, Plecoptera and Trichoptera) biotic indices were used to help detect and measure pollution-related changes to the most pollution-sensitive invertebrates in 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 & Jackson, 1993; Chessman, 1995). However, abundance has been shown to be useful in previous coal mine impact

studies (Clements et al., 2000; Wright & Burgin, 2009a; Belmer et al., 2014; Wright et al., 2017). Similarly, EPT biotic indices were also calculated due to their successful use in previous mine impact studies (Clements et al., 2000; Merriam et al., 2011; Wright et al., 2017). 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 & 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; Hickey & Clements, 1998; Metzeling et al., 2006; Wright & Ryan, 2016) and is one of the most effective and relatively simple biotic index for evaluating ecological impairment from coal mine water pollution (Wright & Burgin, 2009a; Wright et al., 2015). The use of stream ecosystem data is not currently included in the current monitoring program or pollution licence of the coal mine operators or routinely undertaken by the environmental regulator (EPL 726; EPA, 2021).

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, including from coal mines (Merriam et al., 2011; Wright et al., 2015, 2017). 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 amongst samples. All reference site samples were grouped to test differences by two-way analysis of similarity (ANOSIM: Clarke, 1993) between reference sites and sites at the two site categories WSF and WHA downstream of the waste discharges. These multivariate analyses were achieved using the software package PRIMER version 5 (Clarke, 1993).

All water quality, sediment and macroinvertebrate data were tested for homogeneity of variances (using Levene’s test). The normality of distribution and was often skewed. In response, the non-parametric Kruskal–Wallis test was subsequently performed with water, sediment and macroinvertebrate data grouped into three sampling locations (REF, WSF and WHA).

All univariate statistical analysis was performed using IBM SPSS Statistics version 25 (IBM Corporation, 2017).

### 3 Results

#### 3.1 Macroinvertebrates

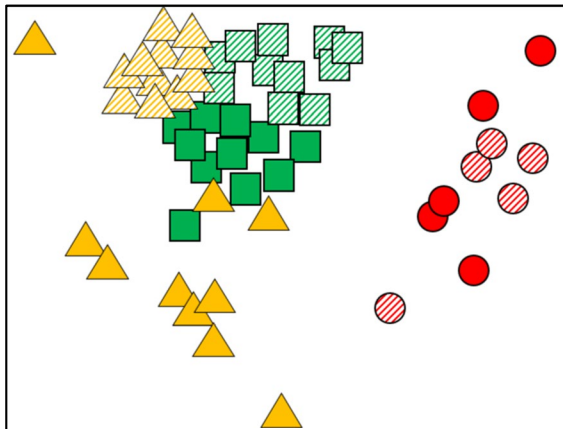
A total of 1,473 individual macroinvertebrates from 33 taxa (mostly families) were collected in the 2020 study; the majority were insects (Table 2).

**Table 2** Macroinvertebrate orders and families present (X = present) at each sampling category in the 2013 and 2020 studies. REF = reference streams, WSF = < 2 km downstream of effluent outfall, WHA = Wollangambe River in World Heritage Area (22 km downstream of effluent outfall). Family name in bold if only detected in 2020

Order	Family	REF13	REF20	WSF13	WSF20	WHA13	WHA20	
Ephemeroptera	Leptophlebiidae	X	X			X	X	
	Coloburiscidae	X	X			X	X	
	Baetidae	X					X	
Plecoptera	Gripopterygidae	X	X	X	X	X	X	
	Eustheniidae	X	X			X	X	
	<b>Notonemouridae</b>		X				X	
Trichoptera	Austroperlidae	X						
	Hydrobiosidae	X	X		X		X	
	<b>Philopotamidae</b>				X		X	
	Hydroptilidae	X	X					
	Ecnomidae	X	X					
	Calocidae	X	X					
	Glossomatidae	X						
	Hydropsychidae	X	X	X			X	
	Leptoceridae	X	X		X	X	X	
	<b>Limnephilidae</b>						X	
	Conosucidae	X			X		X	
	Diptera	Simuliidae	X	X			X	X
		Chironomidae	X	X	X	X	X	X
Ceratopogonidae			X		X		X	
Tipulidae		X		X		X	X	
Empididae		X		X		X	X	
Athericidae						X		
Coleoptera	<b>Curculionidae</b>		X					
	Elmidae	X	X		X	X	X	
	<b>Hygrobiidae</b>		X				X	
	<b>Gyrinidae</b>						X	
Odonata	Scirtidae	X	X		X	X	X	
	<b>Gomphidae</b>		X				X	
Oligochaeta	Aeshnidae		X			X	X	
	Oligochaeta		X	X	X	X	X	
Collembola	Collembola			X	X		X	
Hydrachnidia	Hydrachnidia	X	X	X			X	
Neuroptera	Neurothidae	X					X	
Hemiptera	<b>Veliidae</b>						X	
Megaloptera	Corydalidae	X	X			X		
Lepidoptera	Pyralidae	X						
Total		24	23	8	11	15	28	

Multivariate analysis revealed that the macroinvertebrate community assemblages immediately below the effluent outfall (< 2 km; WSF) remained highly impaired when comparing 2013 and 2020 data with samples collected at reference sites. The nMDS ordination showed that samples from WSF (circle symbols) clustered separately from reference site samples (squares) in 2013 (solid) and again in 2020 (cross-hatched) (Figure 2). The nMDS stress value (0.16) indicated that, in two dimensions, the nMDS was a good to fair representation of the original data (Clarke, 1993) (Figure 2).

Analysis by ANOSIM confirmed the substantial and statistically significant differences in macroinvertebrate community assemblages at reference sites (REF) compared to those collected in the Wollangambe River immediately below the effluent outfall (WSF), in 2013 and again in 2020. The larger  $R$  value from the ANOSIM comparison revealed the ecological dissimilarity between these two site categories in 2013 (REF vs WSF;  $R = 0.457$ ,  $p < 0.001$ ) was slightly greater in 2020 (REF vs. WSF;  $R = 0.515$ ;  $p < 0.001$ ). The macroinvertebrate community at WSF remained ecologically diminished in both 2013 and 2020. In 2013, the mean family richness at WSF was 65.3% lower (3.4, range 3–4 families) than reference sites (9.8, range 8–12 families) (Tables 3



**Figure 2** nMDS plot of macroinvertebrate family data collected from the Wollangambe River in 2013 and 2020. Green squares represent samples from REF = reference streams. Red circles are samples below the outfall WSF = Wollangambe River in State Forest (< 2 km below discharge point). Gold triangles are samples from WHA = Wollangambe River in World Heritage Area (22 km below discharge point). Solid markers represent 2013 and cross-hatched 2020

and 4, Figure 3). In 2020, the reduction in family richness at WSF (mean 3.2, range 2–4 families) compared to reference sites (mean 8.8, range 3–13 families) had an almost identical reduction (63.6%) (Tables 3 and 4). The ecological impairment downstream of the mine outfall (WSF) was also apparent with the lower abundance and family richness of pollution-sensitive EPT macroinvertebrates at this site compared to reference sites, in both 2013 and 2020 (Figure 4). The pollution-sensitive mayfly families Leptophlebiidae and Coloburiscidae were both abundant at reference sites and were never encountered in any samples immediately below the coalmine outfall (WSF) in both surveys but were detected in greater abundance 22 km downstream (WHA) in 2020 than in 2012/13 (Table 5).

In 2020, macroinvertebrate assemblages in the Wollangambe River showed a substantial ecological recovery 22 km downstream of the effluent outfall (WHA), compared to results from 2013. The nMDS ordination showed that the WHA samples (gold triangles) clustered closer to reference site samples in 2020 than in 2013 (Figure 2). ANOSIM confirms that the statistically significant ( $p = 0.001$ ) dissimilarity between macroinvertebrate communities at the most ecologically impaired site (WSF) with the site 22 km the outfall (WHA) was larger ( $R = 0.937$ ) in 2020 than it was in 2013 ( $R = 0.522$ ). In 2020, the macroinvertebrate family richness was 300% greater at WHA (mean 12.8, range 9–19 families) than it was 20 km upstream in the Wollangambe River at WSF (Table 3, Figure 3). Family richness at WHA in 2020 was nearly three times more than it was in 2013 (mean 4.4, range 1–8 families). Macroinvertebrate abundance at WHA in 2020 (mean 84.7, range 43–160 invertebrates/sample) increased by 630% compared to abundance in 2013 (mean 11.6, range 3–38) (Figure 4). The WHA site also recorded more than double the number of EPT families in 2020, compared to 2013 (Tables 3, 4 and 5).

### 3.2 Water and Sediment Quality

Water quality in the Wollangambe River downstream of the entry of mine effluent remained degraded in 2020, but was to a lesser extent than it was in 2012/2013. Electrical conductivity (EC) varied highly significantly according to sample category in 2020, as it did in 2012/13 ( $p = 0.0001$ ) (Tables 3 and 4). The



**Table 3** Summary statistics of 2020 macroinvertebrate biotic index and water quality data indicating *p* value, range and mean (and median) for each variable, according to sampling site category (reference sites vs. WSF sites vs. WHA sites) (see Table 1 for site details). DO = dissolved oxygen.

		REF		WSF		WHA	
Biotic index	<i>p</i> value (K-W)	Range (min.–max.)	Mean (median)	Range	Mean (median)	Range	Mean (median)
Family richness	0.007	3–13	8.8 (9.5)	2–4.	3.2 (4)	9–19.	12.8 (12.5)
Abundance	0.005	28–136	59.9 (49.5)	4–9.	5.4 (5)	43–160	84.7 (85.5)
EPT family richness	0.002	0–7	3.1 (3.5)	0–2	1 (1)	4–7	5.6 (5.5)
EPT abundance	0.009	0–31	10.9 (11)	0–2	1 (1)	9–36.	21 (17.5)
Water quality	<i>p</i> value (K-W)	Range (min.–max.)	Mean (median)	Range	Mean (median)	Range	Mean (median)
Temperature (°C)	0.001	14.2–17.4	16.2 (16.65)	16–19.4	17.5 (17.6)	15.7–18.5	17.2 (17.4)
DO (% saturation)	0.0001	86.3–94	90.7 (91.3)	66.8–90.3	83.4 (86.9)	90.5–92.5	91.7 (91.7)
Turbidity (NTU)	0.087	1.2–6.9	3.3 (2.9)	0.92–49.8	9.6 (2.9)	1.31–3.8	2.4 (2.6)
pH (pH units)	0.0001	4.7–6.9	5.4 (5.1)	6.4–11.9	8.6 (7.5)	4.8–7.1	6.2 (6.4)
EC (µS/cm)	0.0001	24.4–34.8	32.3 (33.3)	253.1–338.1	294.1 (300.1)	131.7–209.3	158.8 (149.9)
Calcium mg/L	0.0001	Bd.	Bd.	19–40	30.1 (33)	9.4–21	13.3 (14)
Potassium mg/L	0.0001	Bd.–0.6	0.35 (Bd.)	Bd.–3	2.4 (2.7)	Bd.–1.9	1.3(1.4)
Sodium mg/L	0.008	3.6–4.7	4.05 (4)	Bd.–3.8	3.2(3.4)	Bd.–4.3	3.3 (3.6)
Magnesium mg/L	0.0001	Bd.–0.6	0.5 (0.5)	Bd.–10	8.3 (9.1)	Bd.–6.2	4.1 (4.3)
Bicarbonate mg/L	0.0001	Bd.	Bd.	Bd.–30	23.2 (24)	Bd.–15	10.6 (12)
Sulphate mg/L	0.0001	Bd.–3	1.6 (2)	44–100	83.7 (86)	30–52	34.9 (38)
Chloride mg/L	0.0001	6–8	6.8 (6)	8–18	14.6 (16)	Bd.–14	9.8 (11)
Aluminium mg/L	0.0001	80–200	124 (100)	40–170	80 (70)	20–110	50.7 (35)
Arsenic µg/L	-	Bd.	Bd.	Bd.	Bd.	Bd.	Bd.
Barium µg/L	0.0001	7–15.	10.3 (8)	23–30	26.5 (27)	16–23	18.9(18)
Cobalt µg/L	0.0001	Bd.–1	0.6 (Bd.)	6–35.	16 (14)	1–4.	2 (1.5)
Copper µg/L	0.591	Bd.–1	0.53 (Bd.)	Bd.–2	0.62 (Bd.)	Bd.–1	0.53 (Bd.)
Iron µg/L	0.006	130–730	310 (210)	56–1200	282.1 (135)	60–440	164.9 (115)
Manganese µg/L	0.0001	25–130	58.5 (32)	80–380	189.1 (160)	32–91	53.2 (39)
Molybdenum µg/L	-	Bd.	Bd.	Bd.	Bd.	Bd.	Bd.
Nickel µg/L	0.0001	Bd.–2	0.6 (Bd.)	23–90	42.6 (33.5)	7–12.	9.2 (9)
Lead µg/L	NA	Bd.	Bd.	Bd.	Bd.	Bd.	Bd.
Lithium µg/L	0.0001	Bd.	Bd.	8–22.	18.6 (20)	5–10.	7.6 (7.5)
Selenium µg/L	-	Bd.	Bd.	Bd.	Bd.	Bd.	Bd.
Strontium µg/L	0.0001	4.7–6.1	5.1 (4.9)	27–66	57.1 (60)	22–43	31.9 (30.5)
Uranium µg/L	NA	Bd.	Bd.	Bd.	Bd.	Bd.	Bd.
Zinc µg/L	0.0001	1–9	3.6 (2)	34–100	60.6 (56.5)	10–14	12.1 (12.5)
Ionic strength (M)			0.0003		0.0041		0.0019
Hardness mg/kg CaCO <sub>3</sub>			4.56		109		50.21

**Table 3** (continued)

	REF	WSF	WHA
Goethite solubility (log Q/K)	5.32	6.39	5.70
Gibbsite solubility (log Q/K)	1.68	0.43	2.07

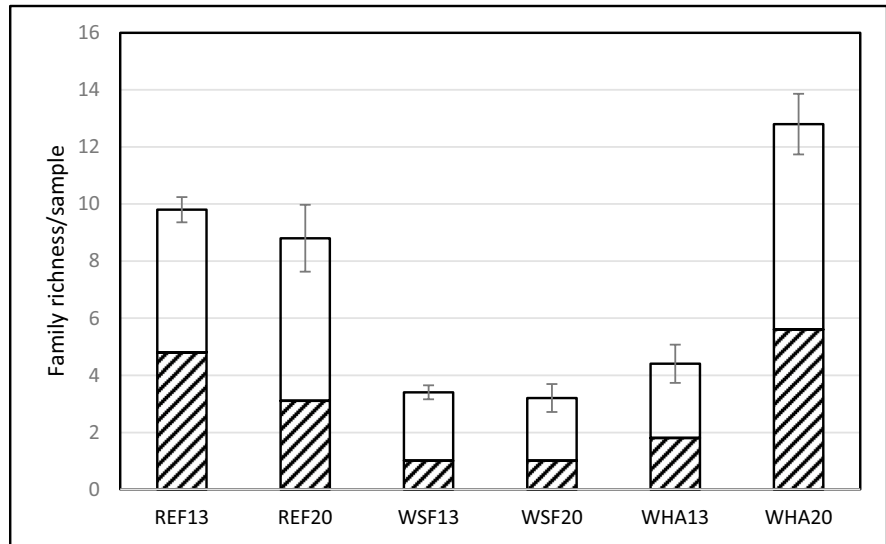
EC = electrical conductivity. Bd. = Below detection limits. K-W= Kruskal–Wallis.

**Table 4** Summary statistics of 2012/2013 macroinvertebrate biotic index and water quality data indicating *p* value, range and mean (and median) for each variable, according to sampling site category (REF reference sites vs. WSF sites vs.

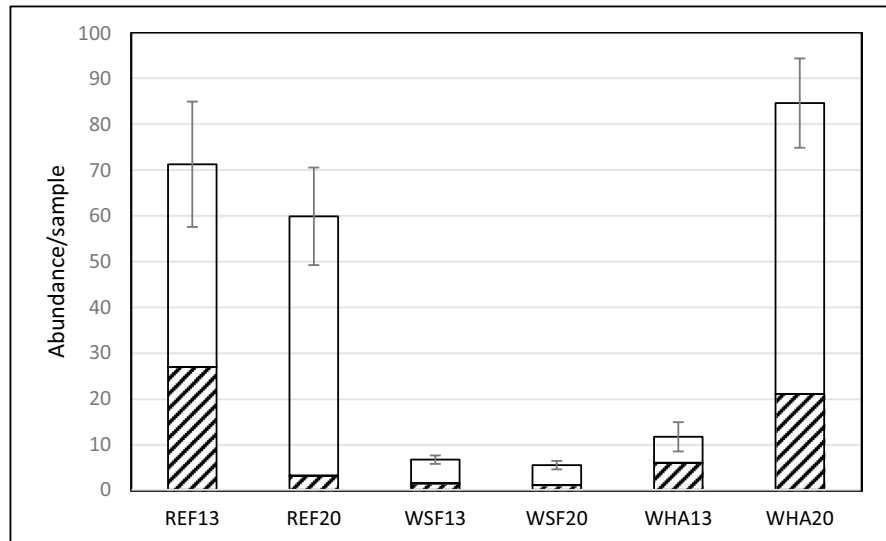
WHA sites) (see Table 1 for site details). DO = dissolved oxygen. EC = electrical conductivity. Bd. = below detectable limits. K-W= Kruskal–Wallis

	REF	WSF	WHA				
Macroinvertebrate biotic index	<i>p</i> value (K-W)	Range (min.–max.)	Mean (median)	Range (min.–max.)	Mean (median)	Range (min.–max.)	Mean (median)
Family richness	0.001	8–12	9.8 (9.5)	3–4	3.4 (3)	1–8	4.4 (4)
Abundance	0.002	27–149	71.1 (60)	4–9	6.6 (7)	3–38	11.6 (9.5)
EPT family richness	0.002	3–8	4.8 (4.5)	0–2	1 (1)	0–3	1.8 (2)
EPT abundance	0.009	7–52	26.9 (22.5)	0–3	1.4 (1)	0–21	5.9 (5)
Water quality	<i>p</i> value (K-W)	Range (min.–max.)	Mean (median)	Range (min.–max.)	Mean (median)	Range (min.–max.)	Mean (median)
Temperature (°C)	0.0001	9.2–18	13.9 (14.6)	12.7–17.8	16.1 (17)	9.4–18.8	15.56 (17)
DO (% saturation)	0.0001	83–99.8	95.3 (96.9)	72.2 (105.9)	94.7 (100.3)	92.4 (124.1)	105.1 (101.9)
pH (pH units)	0.0001	4.9–6.3	5.5 (5.4)	6.9–7.3	7.08 (7.1)	6.4–6.94	6.62 (6.64)
EC (µS/cm)	0.0001	23.5–36	29.4 (28.7)	252–488	378.4 (386)	106.2–326	212.2 (207)
Calcium mg/L	0.0001	Bd.	Bd.	47–63	53.1 (53.5)	9–36	24.4 (25.5)
Magnesium mg/L	0.0001	Bd.	Bd.	11–16	13.6 (13.5)	3–10	6.1 (6.5)
Potassium mg/L	0.0001	Bd.	Bd.	4–5	4.1 (4)	1–3	1.8 (2)
Sodium mg/L	0.81	3–4	3.6 (4)	3–4	3.6 (4)	3–4	3.5 (3.5)
Sulphate mg/L	0.0001	Bd.–7	1.8 (1)	151–201	170.8 (175)	34–122	83.4 (87)
Bicarbonate mg/L	0.0001	Bd.	Bd.	12–22	21.0 (19.7)	2–8	5.58 (6.0)
Chloride mg/L	0.0001	4–8	5.4 (4.5)	4–4	4 (4)	4–6	5.1 (5)
Aluminium µg/L	0.0001	60–300	143.3 (120)	5–70	30 (20)	20–90	46.6 (50)
Copper µg/L	-	Bd.	Bd.	Bd.	Bd.	Bd.	Bd.
Nickel µg/L	0.0001	Bd.–2	0.7 (Bd.)	78–141	101.5 (93.5)	36–48	41.1 (40)
Lead µg/L	-	Bd.	Bd.	Bd.	Bd.	Bd.	Bd.
Zinc µg/L	0.0001	Bd.–7	3.1 (2.5)	94–180	126 (122.5)	40–64	50.1 (48.5)

**Figure 3** Mean total macroinvertebrate family (+/- standard error) and EPT family (cross-hatched) richness collected in 2012/2013 (13) and 2020 (20) from REF = reference sites, WSF = Wollangambe River in State Forest (< 2 km from discharge point), Wollangambe River in WHA = World Heritage Area (22 km downstream of discharge point)



**Figure 4** Mean total macroinvertebrate abundance (+/- standard error) and EPT (cross-hatched) abundance collected in 2012/2013 (13) and 2020 (20) from REF = reference sites, WSF = Wollangambe River in State Forest (< 2 km from discharge point), Wollangambe River in WHA = World Heritage Area (22 km downstream of discharge point)



mean EC in the Wollangambe River immediately below the mine discharge (WSF) dropped by 20% from 378.4  $\mu\text{S}/\text{cm}$  (range 252–488) in 2012/2013 to 294.1  $\mu\text{S}/\text{cm}$  (range 253.1–338.1) in 2020. In 2020, reference sites continued to record the lowest EC (mean 32.3  $\mu\text{S}/\text{cm}$ , range 24.4–34.8) which was a slight increase on the 2012/2013 mean (29.4  $\mu\text{S}/\text{cm}$ , range 23.5–36.0).

The concentration of the metals (zinc and nickel) in water samples varied highly significantly according to sampling site category in both 2020 and 2012/2013 ( $p = 0.0001$ ) (Table 3, 4). Reference sites had similar mean concentrations of zinc in 2012/2013 (3.1  $\mu\text{g}/\text{L}$ ,

range 0–7) and 2020 (3.6  $\mu\text{g}/\text{L}$ , range 1–9) and nickel in 2012/2013 (0.7  $\mu\text{g}/\text{L}$ , range 0–2) and in 2020 (0.6  $\mu\text{g}/\text{L}$ , range 0–2). In both sampling periods, the highest concentrations of zinc and nickel were recorded at WSF in the Wollangambe River below (< 2 km) the mine discharge point. In 2012/2013, the mean zinc and nickel concentrations of 126  $\mu\text{g}/\text{L}$  (range 94–180) and 101.5  $\mu\text{g}/\text{L}$  (range 78–141) were measured at WSF (Table 4). In comparison, in 2020, the mean zinc and nickel concentrations at WSF were about half that recorded in 2012/2013 (Zn 60.6  $\mu\text{g}/\text{L}$ , range 34–100; Ni 42.6  $\mu\text{g}/\text{L}$ , range 23–90) (Table 3). In the Wollangambe River 22 km below the mine, the

**Table 5** Richness and abundance of EPT (Ephemeroptera, Plecoptera and Trichoptera) families recorded in five replicate samples from each of the three site categories in 2012/2013 and 2020. Family name in bold if detected in 2020 but not 2013

EPT family	REF13	REF20	WSF13	WSF20	WHA13	WHA20
Leptophlebiidae	62	30	0	0	20	95
Hydropsychidae	27	10	4	0	0	8
Leptoceridae	13	2	0	1	2	1
Gripopterygidae	126	3	3	1	31	7
Hydrobiosidae	9	37	0	1	0	20
<b>Philopotamidae</b>	0	0	0	1	0	2
Eustheniidae	6	1	0	0	5	16
Coloburiscidae	13	10	0	0	1	18
Baetidae	1	0	0	0	0	35
Hydroptilidae	9	2	0	0	0	0
<b>Notonemouridae</b>	0	14	0	0	0	5
<b>Limnephilidae</b>	0	0	0	0	0	1
Conosucidae	3	0	0	1	0	2
Total EPT Abundance	269	109	7	5	59	210
Total EPT Family Richness	10	9	2	5	5	12

**Table 6** Summary statistics for sediment metal results collected in 2020 indicating *p* value, range and mean (and median) for each metal, by sampling site category (REF refer-

ence sites vs. WSF sites vs. WHA sites) (see Table 1 for site details). Bd. = below detectable limits. K-W= Kruskal–Wallis

2020 sediment chemistry	REF			WSF		WHA	
	<i>p</i> value (K-W)	Range	Mean (median)	Range	Mean (median)	Range	Mean (median)
Aluminium mg/kg	0.006	110–2700	697.5 (175)	2000–8900	4583.3 (3850)	430–7200	4431.25 (6350)
Arsenic mg/kg	-	Bd.	Bd.	Bd.	Bd.	Bd.	Bd.
Barium mg/kg	0.002	1–22.	6 (2.5)	33–130	71.33 (63)	5–75	47.87 (69.5)
Cobalt mg/kg	0.0001	Bd.–2	0.87 (0.5)	310–1200	651.6 (570)	7–35.	12.37 (9)
Copper mg/kg	0.005	Bd.–7	1.81 (0.5)	9–29.	16 (13)	Bd.–14	8.37 (12)
Iron mg/kg	0.001	280–6500	2048.7 (680)	9900–18000	14316.67 (14500)	720–6500	4151.25 (5600)
Manganese mg/kg	0.0001	8–75.	24.25 (12)	2400–13000	6933.3 (6500)	92–310	168.5 (165)
Molybdenum mg/kg	0.0001	Bd.	Bd.	1–8.	3.66 (3)	Bd.	Bd.
Nickel mg/kg	0.0001	Bd.–2	0.75 (Bd.)	640–1400	978.33 (970)	7–23.	12.62 (13)
Lead mg/kg	0.004	Bd.	1.43 (0.5)	5–13.	8 (7)	1–23.	13.37 (19)
Lithium mg/kg	0.0001	Bd.	Bd.	2–50.	20.5 (16.5)	Bd.–2	1 (1)
Selenium mg/kg	-	Bd.	Bd.	Bd.	Bd.	Bd.	Bd.
Strontium mg/kg	0.007	Bd.–5	1.43 (0.5)	5–15.	9 (8)	Bd.–20	11.68 (16.5)
Uranium mg/kg	0.0001	Bd.	0.075 (Bd.)	Bd.–2.2	1.06 (0.85)	Bd.–0.4	0.21 (0.3)
Zinc mg/kg	0.0001	Bd.-7	3.12 (3)	980–3300	2080 (1900)	10–31.	23.87 (27.5)

mean zinc and nickel concentrations were proportionally lower in 2020 than in 2012/2013. In 2012/2013, the mean zinc concentration at WHA was 50.1 µg/L (range 40–64), and in 2020, the mean was 12.1 µg/L (range 10–14). Similarly, in 2012/2013, the mean nickel concentration was 41.1 µg/L (range 36–48), and in 2020, the mean was 9.2 µg/L (range 7–12).

In 2020, the metal content, including zinc and nickel, of waterway sediment samples from all sites was also examined (Table 6). Both zinc and nickel varied significantly, according to sampling category. The mean nickel content was more than 1300 times higher below the mine outfall at WSF (978.3 mg/kg, range 640–1400) than at reference sites (REF

0.75 mg/kg, range 0–2). The nickel content in Wollangambe River sediment 22 km downstream of the mine (WHA mean 12.62 mg/kg, range 7–23) was 98% lower than at WSF. The same pattern was also found for the mean zinc content of river sediment with 3.12 mg/kg (range 0–7) at REF, rising by more than 600 times to 2080 mg/kg (range 980–3300) at WSF and falling to 23.87 mg/kg (mean 10–31) at WHA (Table 6).

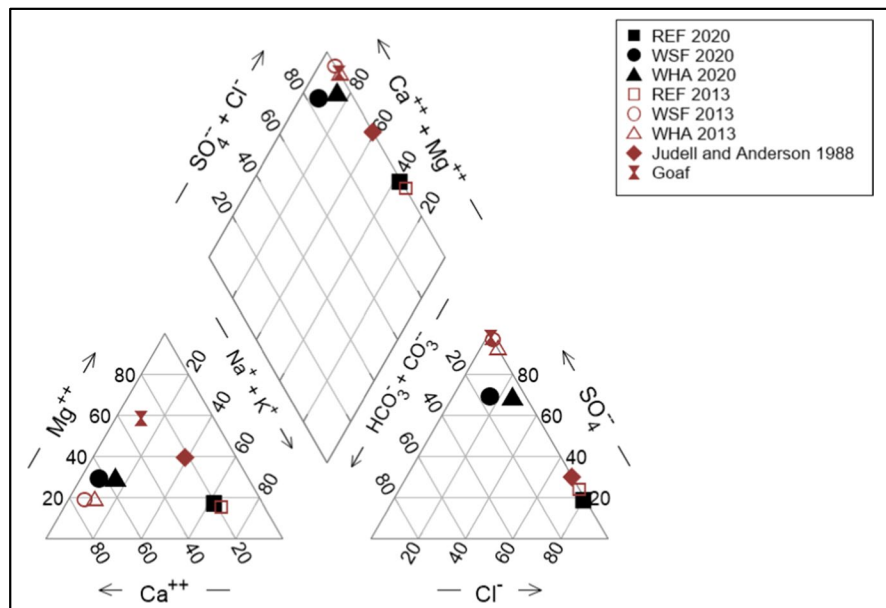
The ionic composition of the Wollangambe River, below the effluent outfall, remained very different to that at reference sites. In both 2012/2013 and 2020, the dominant ions at reference sites were sodium (2012/2013 mean 3.6 mg/L, range 3–4; 2020 mean 4.05 mg/L, range 3.6–4.7) and chloride (2012/2013 mean 5.4 mg/L, range 4–8; 2020 mean 6.7 mg/L, range 6–8) (Tables 3 and 4). Immediately downstream of the outfall (WSF), the dominant ions were calcium (2012/2013 mean 53.1 mg/L, range 47–63; 2020 mean 30.1 mg/L, range 19–40) and sulphate (2012/2013 mean 170.8 mg/L, range 151–201; 2020 mean 83.7 mg/L, range 44–100) (Table 3, 4). Twenty-two km downstream of the outfall (WHA), calcium was recorded at (2012/2013 mean 24.4 mg/L, range 9–36; 2020 mean 13.3 mg/L, range 9.4–21) and sulphate (2012/2013 mean 83.4 mg/L, range 34–122; 2020 mean 34.9 mg/L, range 30–52) both remaining the dominant ions. These ionic composition changes in the Wollangambe River can be characterised as a

shift away from a dilute sodium chloride-dominated system to a colliery effluent-modified calcium sulphate-dominated system (Figure 5).

### 4 Discussion

This current research reveals that two decades after the Greater Blue Mountains was listed by UNESCO as a natural World Heritage site, progressive tightening and more effective environmental regulations are reducing the severity of coal mine pollution in the Wollangambe River. The improvement in water quality and ecological health is most apparent 22 km downstream of the mine discharge, when comparing results from the current 2020 study with an earlier study in 2012/2013 (Belmer et al., 2014; Wright et al., 2017). Water and sediment in the Wollangambe River immediately downstream of the effluent discharge (< 2 km) remains contaminated and contributes to impairment of macroinvertebrate communities. Decades of disposal of metal-enriched colliery wastes have accumulated a reservoir of nickel and zinc within Wollangambe River sediments, which are now a pollution legacy that will continue to mobilise contaminants and impair river water quality and ecological health for an unknown period.

**Figure 5** Major anion and cation ternary diagram (% meq/kg) represents the ionic composition similarity that water samples collected in the three categories (REF, WSF, WHA) in 2020 and 2013. Comparison is also made with mine water samples and *goaf* water samples collected within Clarence Colliery by Judell and Anderson (1988)





#### 4.1 Ecological Recovery in the World Heritage Area 22 km Below the Mine

The improving ecological health of the Wollangambe River at WHA, 22 km below the mine outfall, in 2020 was shown by the increased taxonomic richness and abundance of pollution-sensitive Ephemeroptera, Plecoptera and Trichoptera (EPT) order invertebrates, compared to results collected in 2012/2013. EPT invertebrates have a worldwide reputation as being sensitive to water pollution (Lenat & Penrose, 1996). Diminished abundance of EPT invertebrates in waterways contaminated by coal mines was demonstrated in the Rocky Mountain (Colorado, USA) streams polluted with heavy metals (Clements et al., 2000). This current study recorded a steep (256%) increase in the abundance of EPT invertebrates, compared to 2012/2013 results, in the Wollangambe River, deep within the World Heritage Area, 22 km below the mine outfall (WHA).

One of the most abundant EPT families collected in this study was the pollution-sensitive Leptophlebiidae family (Chessman, 1995; Belmer & Wright, 2021). The family Leptophlebiidae is very common in nearby uncontaminated streams in the Grose River but was not collected in the site most contaminated by zinc-enriched drainage from Canyon Colliery flowing into the Grose River via Dalpura Creek (Wright & Burgin, 2009a). The family Leptophlebiidae has a well-known intolerance of many forms of poor water quality including sewage and metal pollution (Chessman and McEvoy, 1998). This family was absent immediately below a treated sewage outfall on Hat Hill Creek (Wright & Burgin, 2009a). In this current study, no individuals from the Leptophlebiidae family were detected in 2012/2013 or in 2020 in the heavily contaminated Wollangambe River < 2 km below the mine outfall (WSF). However, at the site 22 km downstream of the colliery discharge point, Leptophlebiidae has shown signs of recovery, reporting an almost four times increase in abundance between 2012/2013 and 2020.

#### 4.2 Water Quality Improvement

Zinc and nickel emerged as pollutants of most ecological concern in the 2012/2013 study in the Wollangambe River downstream of the mine outfall (Belmer et al., 2014; Wright et al., 2017). The mean

concentrations of nickel and zinc at all sampling sites in the Wollangambe River below the mine outfall in both 2012/2013 and 2020 were above concentrations considered ecologically hazardous, according to Australian water quality guidelines for protection of aquatic species (ANZECC, 2000). Trigger values for freshwater streams to achieve 99% level of ecosystem protection are  $\leq 8 \mu\text{g/L}$  for nickel and  $\leq 2.4 \mu\text{g/L}$  for zinc (ANZECC, 2000). In 2012/2013 at the site < 2km below the mine outfall (WSF), mean zinc and nickel concentrations (nickel 101.5  $\mu\text{g/L}$ ; zinc 126  $\mu\text{g/L}$ ) were many times higher than hazardous levels for aquatic species. However, when comparing mean concentrations between 2012/2013 and 2020, there has been a considerable reduction of 50% or more (WSF, nickel 42.6  $\mu\text{g/L}$ ; zinc 60.8  $\mu\text{g/L}$ ). This can be attributed to the change in operation of the mine as a result of the modified environmental pollution licence. At the furthest downstream site (WHA), 22 km below the discharge point, the concentrations of both metals in 2020 also reported lower levels between the 2012/2013 and 2020 periods. While 2020 data for WHA indicate that concentrations are still exceeding ANZECC guidelines (mean nickel 9.2  $\mu\text{g/L}$  and mean zinc 12.1  $\mu\text{g/L}$ ), they are both about 75% lower than in 2012/2013 and are trending towards the safer trigger levels.

Although exceeding ANZECC (2000) guidelines for ecological protection, the concentrations of zinc and nickel measured in the Wollangambe River (2012/2013 and 2020), immediately below the mine outfall, are relatively low but are broadly comparable to a similar study. Herbst et al. (2018) investigated acid mine drainage and degradation of stream macroinvertebrate communities in Sierra Nevada streams (California, USA). That study detected zinc concentrations (c. 40  $\mu\text{g/L}$ ) at the most degraded waterway in their study (Herbst et al., 2018). This was at least 30% less than the zinc concentration we measured in the Wollangambe near the mine outfall in 2020. However, nickel concentrations at the most ecologically degraded Sierra Nevada site (Herbst et al., 2018) (67–223  $\mu\text{g/L}$ ) were often greater than the maximum concentration (141  $\mu\text{g/L}$ ) we encountered in 2012/2013 at the most ecologically impaired site in the Wollangambe River. Indiana (USA) waterways receiving coal mine effluent reported nickel concentrations (c. 110  $\mu\text{g/L}$ ) similar to Wollangambe in 2012/2013, and zinc concentrations (c. 310  $\mu\text{g/L}$ ) were more than

double the highest mean concentration (126 µg/L) recorded in the Wollangambe in 2012/2013.

The mechanisms by which metals (such as zinc and nickel) harm aquatic life in rivers and streams are not fully understood. Researchers have conducted ecotoxicology investigations of metals on the larval stage of aquatic insects in controlled laboratory environments that often encounter greater tolerance of insects to elevated concentration of metals than are encountered in field studies (Clements et al., 2013). It is plausible that the larval life stage of aquatic insects could be more tolerant to elevated metals. For example, a study of benthic larvae and adult insect emergence from metal (zinc and lead)-contaminated lakes in northern Sweden concluded that aquatic insect metamorphosis was a vulnerable life stage to elevated metals and could be a survival bottleneck for all insect taxa (Lidman et al., 2020).

The lower zinc and nickel concentrations measured in the water column of the Wollangambe River in 2020, compared to 2012/2013, can be attributed to Clarence Colliery releasing effluent with reduced loads of zinc and nickel, in accordance with their NSW EPA licence (EPL, 726; EPA, 2021). During the 2012/2013 study, the NSW EPA pollution licence permitted Clarence Colliery to release effluent with a zinc concentration of up to 2500 µg/L (EPA, 2021; Notice 1502867). According to results of monthly samples collected and reported by Centennial Coal between December 2012 and May 2013, the colliery effluent, during that period, had zinc concentrations ranging from 117 to 362 µg/L (Centennial Coal, 2021). The concentration of zinc in colliery effluent has since fallen steeply in the following 7 years. In January 2020 (EPL 726; EPA 2021), permitted zinc concentrations of less than 8 µg/L and the colliery reported zinc of < 5 µg/L (twice) and 8 µg/L in the three monthly samples from October to December 2020 (Centennial Coal, 2021). Nickel was not included in the NSW EPA licence for Clarence Colliery, and no nickel results were reported by the colliery in 2012 and 2013. Nickel was first specified in the colliery NSW EPA pollution licence (EPL 726) in 2017 at a maximum concentration of 11 µg/l (EPL 726, EPA 2021; Notice # 1590599). During October to December 2020, it was reported at 3, 7 and 8 µg/l (Centennial Coal, 2021).

The concentrations of zinc and nickel in water samples collected near the mine discharge (WSF,

< 2 km mine outfall) in this study (2020) were also consistent with results reported by Centennial Coal at similar sample location. The colliery is required by EPL 726 to conduct monthly testing and report a suite of pollutants in the Wollangambe River, downstream of the mine waste discharge. Over a 6-month period (July to December 2020), Centennial Coal reported nickel concentrations ranging from 22 to 38 µg/l (Point 9, EPL 726; EPA 2021) at this site on the Wollangambe River, 450 m upstream of the WHA boundary (Centennial Coal, 2021). Zinc concentrations at the same site ranged from 22 to 58 µg/l in the same 6-month period (July to December 2020; Centennial Coal, 2021).

#### 4.3 Legacy Pollution from Reservoir of Contaminated River Sediments

The risk of metal-contaminated sediments in the Wollangambe River becoming a reservoir of legacy pollutants was predicted by Lorenzelli et al. (2018). Lorenzelli et al. (2018) investigated Wollangambe River sediments in laboratory leaching experiments at different pH treatments. His study found that mobilisation of nickel and zinc from sediment to the water column was highly sensitive to solution pH. For example, the resulting zinc concentration more than 25 times higher at pH 6 compared to pH of 8 (Lorenzelli et al., 2018).

We hypothesise that the higher nickel and zinc concentrations measured in the river (< 2 km) downstream of the mine discharge compared to concentrations reported in the colliery effluent is due to the remobilisation of zinc and nickel from the metal-enriched river sediments. Testing of river sediment samples from all sites in this 2020 study showed that the river sediments in the Wollangambe River (WSF, < 2 km mine outfall) had the greatest metal content, particularly for zinc and nickel. Remobilisation of zinc and nickel from the sediment may occur as part of adsorption phenomena in response to increasing acidity or potentially as a result of Ostwald ripening or iron (oxy)hydroxides due to sediment aging. We suggest the former is important where pH change may decrease from mean pH of 8.6 at WSF to 6.2 at WHA following the findings of Lorenzelli et al. (2018). The ripening of iron (oxy)hydroxide sedimentary precipitates will also contribute to the release of zinc and nickel where precipitates may incorporate these metals (Yan et al., 2021).

The river sediments in the Wollangambe River (WSF, < 2 km mine outfall) had a mean zinc content of 2080 mg/kg and mean nickel content of 978 mg/kg. These results are environmentally hazardous as they exceeded the default Australian sediment quality guidelines for both zinc and nickel (ANZECC, 2019). The sediment samples for this site (WSF, < 2 km mine outfall) exceeded the default guideline value for zinc (< 200 mg/kg) by ten times and the default guideline value for nickel (< 21 mg/kg) by more than 45 times. Such zinc and nickel concentrations appear unusually highly elevated for river sediments exposed to coal mine waste. For example, a survey of sediment samples from a Canada-wide survey of waterways exposed to coal mines or coal-fired power stations reported nickel < 50 mg/kg and zinc < 200 mg/kg (Cheam et al., 2000). A similar study conducted in China also reported similar nickel (< 50 mg/kg) and zinc (< 200 mg/kg) results from river sediments in a region affected by coal mine activity (Ji et al., 2018).

Many international studies have reported that legacy metal contamination of river sediments can cause long-term remobilisation of metals into the water column for many decades. Research that has investigated zinc and nickel contamination of river sediments associated with remobilisation and impairment of river water quality shares many similarities to the current study. A major difference is that most zinc and nickel remobilisation studies are more frequently associated with metalliferous mining and industrial activity than coal mining alone. For example, an investigation conducted in southeast France examined a region where a combination of several centuries of metal (gold, lead, tin) mining, as well as coal mining and other industrial activity, caused metal enrichment of sediments that causes long-term remobilisation of metals and contributes to degraded water quality in the Gardon and other rivers (Resongles et al., 2014). The zinc content of most sediments in the Gardon study were < 200 mg/kg with a maximum reported of 1197 mg/kg (Resongles et al., 2014). In comparison, Wollangambe sediments at the most contaminated site (WSF, < 2 km mine outfall) in the current study ranged between 980 and 3300 mg/kg and are comparable, but slightly lower, to the range of zinc sediment results reported from the Shiine River catchment in Japan which ranged between 87 and 7490 mg/kg (Shikazono et al., 2008). Similar to the Wollangambe River, the Shikazono et al. (2008) study

concluded that zinc, and other metals, were mobilised from metal-enriched river and catchment sediments and contributed to the long-term increase of the zinc concentration of river water. An investigation of river sediments in Italy, from an abandoned copper mine, reported nickel content ranging from 192 to 1118 mg/kg (Consani et al., 2017). In our study, the nickel content in the most contaminated site (WSF, < 2 km mine outfall) was greater, ranging from 640 to 1400 mg/kg.

#### 4.4 Mining Impacts on Protected World Heritage Areas

This is not an isolated case study of contamination from mining activities adversely affecting UNESCO World Heritage Areas. Gold and uranium mining and associated AMD contamination of surface and groundwater water have threatened the integrity of the ‘Cradle of Humankind’ World Heritage Site (Krugersdorp, South Africa; UNESCO, 2021b; Durand et al., 2010). This site has a sensitive karst system that contains amongst the world’s most important paleontological and archaeological sites (UNESCO, 2021b). Another World Heritage Area that has been contaminated by mining activity is Bathurst Harbour in south west Tasmania, part of the Tasmanian Wilderness World Heritage Area (Saunders et al., 2013; UNESCO, 2021c). Bathurst Harbour is an isolated and protected estuary that was considered to be near-pristine, but a study of *c.*350 year history of sediment cores has detected impairment of algal diatom communities and increased metal concentrations since 1870 (Saunders et al., 2013). The source of the elevated metals (including arsenic, antimony, copper and zinc) in Bathurst Harbour sediment has been attributed to historic small local mines and fallout originating from distant mining activity in other regions of Tasmania (Saunders et al., 2013).

Clarence Colliery is located close to the boundary of, and discharges into the headwaters of, one of the most highly protected conservation areas and rivers in Australia. Prior to its listing as a World Heritage site in November 2000, the Commonwealth Government dismissed the threat of coal mines to the conservation value of the Blue Mountains area. This was explained in the Australian Government’s proposal for the Blue Mountains area to be inscribed by UNESCO on the World Heritage list where it claimed that ‘...some

coal mine operations occur nearby, but do not affect the water catchments that drain to the area' (Australian Government, 1998). Additionally, the Australian Government made the following assurance in the Blue Mountains World Heritage nomination: '... all mining operations are subject to strict development control and pollution licencing by the New South Wales Environmental Protection Agency' (Australian Government, 1998). Such statements of the optimism of the regulatory process, clearly, lack an evidence-based rigour and, while there are positive outcomes with the most recent EPL for Clarence Colliery, remain silent with respect to closed and other mines within the GBMWhA.

Within the Greater Blue Mountains World Heritage area, and surrounding lands that border the area, pollution from coal mines is not contained to just the Clarence Colliery (UNESCO, 2021a). The nearby Canyon Colliery continues to release untreated mine drainage from its flooded underground workings without any remediation or pollution licencing requirements since its closure in 1997. The contaminated drainage continues to pollute the Grose River and impair its ecological condition within the conservation area and World Heritage boundary (Wright & Burgin, 2009a, b; Wright et al., 2011; Price & Wright, 2016; Belmer & Wright, 2020; IUCN, 2020). There is also substantial water and sediment pollution from two active coal mines and several closed coal mines in the adjoining Coxs River catchment which also flows into the GBMWhA (Birch et al., 2001; Battaglia et al., 2005; Belmer & Wright, 2020).

#### 4.5 Tighter Environmental Regulations of Clarence Colliery

For nearly 25 years of operation (1980 to 2004), the NSW EPA's environmental pollution licence enabled the Clarence mine to discharge two hazardous metals (zinc or nickel) in its waste discharge into the Wollangambe River without any limitations (EPL 726, EPA 2021). More recently there has been an incremental change to the pollution licencing process that reflects a newer regulatory risk-based approach of the environmental regulator (NSW EPA, 2013), albeit, very gradually. While the mine now discharges much lower concentrations of many pollutants, the enrichment of zinc and nickel content in river sediments remains an accumulated legacy. It appears likely that

the reservoir of zinc and nickel will continue to remobilise metals into the water column of the river and have ongoing ecological impacts for decades (Shikazono et al., 2008; Resongles et al., 2014).

This current case study highlights a shortcoming of the pollution licencing system in NSW, notably with regard to delays and slow modification to the coal miner's pollution licence to adequately protect the receiving environment. Moreover, this case study also highlights a lack of intervention by the Australian Commonwealth Government with respect to their World Heritage obligations as signatories to the United Nations World Heritage Convention in 1972 (UNESCO 2021d) and subsequent directions to the state environment agency.

## 5 Conclusion

This current study represents a detailed assessment of water and sediment quality and the ecological health of a high conservation-value river that was impaired by coal mine effluent. Encouragingly, it documents improved water quality and ecological health in response to improved environmental regulation. There are three notable points. First, this is a mining operation within an internationally significant environmentally sensitive area that has a long history of environmental impact. Second, it sheds a positive light on how a risk-based pollution licencing system can lead to positive environmental outcomes through the progressive tightening of discharge limits. Third, despite recent and progressive regulatory action, legacy pollution within the sediments near the mine discharge will continue to contribute to the pollution of the environment for decades to come. This legacy pollution should serve as a reminder to environmental regulators of the critical role pollution licencing can play in proactively protecting the natural environment.

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**Data Availability** Data will be provided on request to the corresponding author.

**Code Availability** Not applicable.

## Declarations

**Conflict of Interest** The authors declare no competing interests.

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