Berrima Colliery
Performance Monitoring Program
Scientific Report
Environment Protection Licence 608

28 February 2019

Prepared By:
International Environmental Consultants Pty Limited
“Longmead”
700 Wombeyan Caves Road
High Range NSW 2575

Table of Contents

Appendices

1. Abstract

Berrima Colliery is an underground coal mine located in the Southern Highlands of NSW. Originally commencing in the 1870's the mine operated continuously between 1926 and 2013. Since 2013, the mine has been in the process of final closure. While operating, the mine pumped water to surface for domestic use at the pit top as well as supplying the nearby Village of Medway. Excess water was discharged into Wingecarribee River via a licensed discharge point. Prior to discharge, the underground water was pumped and settled multiple times which reduced, though not eliminated a range of minerals that occurred naturally in the groundwater. Discharge from the mine has occurred for approximately 90 years.

During the final closure process, the mine was allowed to flood and free drain into the Wingecarribee River via the existing licensed discharge point. Some water is still pumped to the surface but is no longer supplied to residents in Medway. The rate of discharge has averaged 2.5 ML per day which is slightly below the historic average.

The water quality draining from the flooded mine had elevated Iron, Manganese, Nickel and Zinc, lower pH and higher dissolved solids. Consequently, an underground treatment system was installed which involved aeration and pH correction using limestone aggregate followed by settlement to remove precipitated minerals. The purpose of the treatment system was to bring the water quality discharge back to the historic discharge quality.

This report compiles four separate studies into the impacts on the river, namely water quality, aquatic ecology, ecotoxicology and sediment analysis. These results can then be used to develop further strategies necessary to provide long term certainty of the potential impacts of the mine following permanent closure.

2. Introduction

2.1 Purpose

This report has been prepared in response to Special Condition E3.1 of Environment Protection Licence (EPL) 608.

2.2 Scope

Special Condition 8 of EPL 608 requires Berrima Colliery to develop and implement an action plan to prevent, control, abate and mitigate pollution to the Wingecarribee River. Special Condition 8 includes a set of sub-conditions (E1 to E3) which require assessment of background water quality, development of alternative water uses, specifies the scope of the underground water treatment methods, maintenance of the treatment system, performance monitoring and reporting requirements.

This report provides the results of the underground water treatment system available for the period ending 31st December 2018.

2.3 Study Area

The overall study area consists of Consolidated Coal Lease 748 shown on Plan 1 in Appendix A, but centres on the underground mine workings rather than the two surface sites referred to as the Pit Top and Loch Catherine. These sites were used for administration, coal handling and storage and do not contribute to the discharge of the mine.

The specific study area consists of the mixing zone of the mine discharge within the Wingecarribee River. This extends from the licensed discharge point downstream for a distance of approximately 6 km. Reference sites upstream of the discharge point and in the Medway Rivulet tributary form ambient sites for comparison. Historical upstream and downstream sample locations are also included to provide a more comprehensive data set on river water quality data.

2.4 Aims and Objectives

To primary aim of the work is to assess the impacts of the water discharge from the mine on the receiving waters of the Wingecarribee River using the risk based methodology contained in the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC and ARMCANZ 2000). As the mine has historically discharged into the river, there is the need to assess the changes since the water quality deteriorated using the historic data as a baseline. The objectives specified by the EPA in Condition E3.1 of EPL608 require:

> *"To assess the gradient changes in composition and abundance of in-stream biota downstream of Berrima Colliery's adit discharge to the Wingecarribee River. To assess*

> *changes over time in the following installation of the water treatment system. To determine whether water quality in the Wingecarribee River at Biloela is less than trigger values for primary industries and recreational water quality and aesthetics (ANZECC 2000)."*

The hypothesis put forward by the EPA in Condition E3.1 is as follows:

> *"That the abundance and composition of aquatic biota will become more similar to reference sites following commissioning of the required water project. A specific bioindicator target is for % EPT at sites downstream of the discharge (Point 4, 5 and 6) are to be statistically similar to reference sites. The EPT index is used to calculate the relative abundance of pollution sensitive macroinvertebrates of the Ephemeroptera, Plecoptera and Trichoptera Orders. (Wright & Ryan, 2016)."*

This report covers the first 12 months of the Performance Monitoring Program. The final Scientific Report will be prepared by 28th February 2020.

2.5 Methods

There are several components to this study, namely:

- ☐ Surface water quality within the receiving waters of the Wingecarribee River. This includes both the mixing zone and nominated reference sites.
- ☐ Groundwater quality within the mine workings prior to release into the Wingecarribee River. Although the water is naturally occurring and the discharge longstanding, the recent change in water quality following cessation of mining has created the need to undertake investigations into removing higher mineral content to better match long term discharge quality.
- ☐ Aquatic ecology studies within the receiving waters, including the mixing zone and nominated reference sites.
- ☐ Ectoxicological investigations to determine changes in inhibitors within the mixing zone compared to nominated reference sites.
- ☐ Sediment analysis along the river to determine rate of transport, effects of geology and the ultimate fate of minerals discharged within the mixing zone.

The above studies were undertaken during the calendar year 2018, however a similar set of studies were undertaken between 2011 and 2012, while ambient water quality within the Wingecarribee River has been undertaken continuously since 2010. An ANZECC assessment was completed for Berrima Colliery in February 2013 which provides a baseline to compare with studies undertaken in 2018. Details of each study is provided in the following chapters.

The methods used in this study follow the risk-based process described in ANZECC (2000) and the Guidelines in ANZECC (DEC 2006). Details of the methods for each study component is separately provided in each of the following chapters, however there are some guiding principles provided by ANZECC (2000) which are observed.

In Section 2.2.1.9, ANZECC 2000 page 2-17 states:

> *"The Guidelines have not been designed for direct application in activities such as discharge consents, recycled water quality or stormwater quality, nor should they be used in this way. (The exception to this may be water quality in stormwater systems that are regarded as having some conservation value.) They have been derived to apply to the ambient waters that receive effluent or stormwater discharges, and protect the environmental values they support. In this respect, the Guidelines have not been designed to deal with mixing zones, explicitly defined areas around an effluent discharge where the water quality may still be below that required to protect the designated environmental values. As such, the application and management of mixing zones are independent but very important processes."*

The ANZECC guidelines are a risk-based process designed to allow for the development of appropriate triggers based on ambient water quality and ecological investigations. The default water quality concentrations quoted in ANZECC should not be taken as 'pass' or 'fail' criteria. As stated in Using the ANZECC Guidelines and Water Quality Objectives in NSW (DECC 2006):

> *"Trigger values are fundamental to using the ANZECC guidelines. The trigger values for different indicators of water quality may be given as a threshold value or a range of desirable values. Trigger values are conservative assessment levels, not pass/fail' compliance criteria. Local conditions vary naturally between waterways and it may be necessary to tailor trigger values to local conditions or 'local guideline levels'. The guidelines provide a process for refining the trigger values and these protocols should always be followed.*

The ANZECC methodology outlines the following sequence to be followed to determine appropriate trigger values:

- Define the water body including scientific information, monitoring data and ecosystem type classification (ANZECC 2000, Section 3.1.2).
- Determine environmental values to be protected.
- Determine level of protection (ANZECC 2000, Section 3.1.3).
- Identify environmental concerns such as toxic effects, nuisance aquatic plant growth, maintenance of dissolved oxygen and changes in salinity.
- Determine major natural and anthropogenic factors affecting the ecosystem.
- Determine management goals.
- Determine a balance of indicator types (ANZECC 2000, Section 7.2.1).
- Select indicators relevant to concerns and goals.
- Determine appropriate guideline trigger vales.
- Determine environmental values.
- Determine Site Specific Trigger Values.

☐ Determine effects on ecosystem-specific modifying factors including biological assessment, physical and chemical stressors, toxicants and sediments.

Given the nature of this study, some components have been developed separately as part of regulatory authority responsibilities under the mining lease CCL748 (Mining Act) and Environment Protection Licence EPL608 (Protection of the Environment Operations Act). The two principle authorities are the Environment Protection Authority (EPA) and the Department of Planning and Environment – Resources Regulator.

In terms of study methodology, the EPA has specified that the study should *“determine whether water quality in the Wingecarribee River at Biloela is less than trigger values for primary industries and recreational water quality and aesthetics”* (EPL608 Condition E3.1). This is an appropriate environmental value. The EPA has also specified some aquatic ecology methods and has undertaken its own ecotoxicological testing which is included in this study along with data obtained from earlier studies when the mine was operational for comparison.

3. Water Quality

3.1 Introduction

The trial underground treatment system seeks to emulate the original underground water management system within the confines of the non-flooded mine workings. When operating, underground water was collected at several points within the mine and either pumped or allowed to gravity flow into a large sump at the lowest point in the mine (400 Panel Main Sump). From here the water was pumped into the 4a/4b Sump, then K Mart Sump then finally the Pit Bottom Sump. From here the water overflowed into the old mine workings and discharged into the Wingecarribee River at the licensed discharge point via the drain adit.

The mine water would generally travel up to 4 km underground and passed through at least 3 separate aeration and settlement processes prior to discharge. Aeration was provided by pumping and gravity feeding the water along channels. This process also maintained the pH at or slightly above neutral while the large sumps provided time for settlement. This process was able to remove the majority of iron and a proportion of manganese and other minerals from the water prior to discharge.

As part of the closure process, the first three underground sumps were allowed to flood and are no longer accessible. Only the Pit Bottom Sump and the remaining dry mine workings can be utilised. The current underground treatment system involves pumping from the edge of the flooded workings along 400 Panel to 3 North Panel where it is passed through a limestone bed and weir arrangement to increase pH. The water then passes through old workings into the Pit Bottom Sump for settlement prior to discharge via the Drain Adit. The use of limestone and active aeration is necessary given the much smaller available area underground compare to the original water management system.

The underground trials of the treatment system commenced in early February 2018. The quality of the water at the end of the limestone channel has shown an increase of 0.5 pH units and much improved dissolved oxygen concentration. This has seen a significant reduction in dissolved iron concentration as well as reduced concentrations of manganese and nickel. There has been an overall improvement in discharge quality however there is still a variance in the current water quality compared to historic discharge water quality.

The underground treatment system will progress and may see variations to increase surface contact time with limestone and greater settlement time to remove metal precipitates. The system will also vary as the mine progresses toward the final closure. This will include the installation of underground bulkheads to determine strata permeability and to facilitate the determination of final closure arrangements.

3.2 Methods

On a monthly basis, water samples are taken from five sites within the Berrima Colliery underground workings (including the Adit Discharge Point), and from eight sites along the Wingecarribee River (including reference sites) as part of the Performance Monitoring Program. Three additional ambient river sites are sampled bi-monthly. Water samples are collected in plastic bottles with required preservatives and taken to ALS Environmental Laboratories on the same day for analysis. Chain of Custody documentation is prepared, and records kept of the time of sampling, location, person collecting the sample, temperature, pH and conductivity of the sample at collection, time received at the laboratory, temperature of the sample received at the laboratory and cross checking of analytes to be tested. The analytes tested include but are not limited to: Total and dissolved Metals, Alkalinity, Dissolved Cations, Ionic Balance, Nitrogen, pH, Conductivity and Dissolved Oxygen.

3.2.1 Underground Monitoring Locations

Five sites within the underground water treatment are routinely sampled. These sites area as follow:

- ❐ **400 Panel D12 Sump (D12 Sump)**: this water is collected via a pump and pipeline from the flooded mine workings and represents the beginning of the treatment system. The location of the sump is in 400 Panel D Heading at 12 cut-through.
- ❐ **3 North B1 Discharge**: once being pumped from the D12 Sump, the water is transferred via a pipe to a temporary holding tank (referred to as the Fish Tank) where further aeration occurs, then pumped again via a pipe to the 3 North Discharge site. The water tested at 3 North B1 Discharge represents treatment with aeration only.
- ❐ **400 Panel C5 and 400 Panel C8**: following discharge at the 3 North B1 Discharge location, the water commences passive treatment as it flows down dip over the limestone channel and weir arrangement constructed along the 3 North Travelling Road. The treated water then flows through the goaf between the 3 North treatment area and 400 Panel where it is monitored at 5 cut-through and 8 cut-through (referred to as 400 C5 and 400 C8 respectively). This water represents water treated with aeration and passive contact with limestone prior to draining via gravity through to the drain adit via the Pit Bottom sump.
- ❐ **Drain Adit Discharge**: This sampling site is indicative of the water quality leaving the mine.

3.2.2 Performance Water Quality Monitoring Program

On 21st December 2017, the EPA varied EPL 608 to include additional conditions in relation to water treatment and management prior to discharge under a Performance Monitoring Program. The Resources Regulator has also requested additional monitoring activities during the closure process.

A key component of the Performance Monitoring Program is the resultant water quality discharged into the Wingecarribee River and the assessment of any changes in water quality occurring within the mixing zone. The program includes additional monitoring locations downstream of the discharge point as well as reference sites just upstream of the discharge point. The program also includes the historic monitoring locations both upstream and downstream of the mine discharge for comparison purposes.

Water quality monitoring is undertaken monthly within the Wingecarribee River both upstream of the adit discharge and at various locations downstream. A monthly sample is also taken from the Medway Rivulet which is a tributary of the Wingecarribee River with the confluence approximately 2km downstream of the mine adit. The adit discharge water is also routinely sampled as part of the monitoring program. The program commenced in January 2018 and will continue at this stage until the end of 2019. The sampling design consists of three groups as described in the following sections and shown on the attached plan.

3.2.2.1 Discharge Monitoring Sites - Near

These sites represent the mixing zone within the Wingecarribee River as well as the discharge water from the drain adit and are numbered as follows:

- Site 1: Mine Adit - Naturally occurring groundwater is captured in the underground workings and is discharged into the Wingecarribee River. The monitoring point is referred to as the V Notch Weir (EPL Point 4).
- Site 3: WR 300 DN – This site is located in the Wingecarribee River approximately 300m downstream of the confluence with the mine discharge water.
- Site 4: WR 1km DN - Wingecarribee River approximately 1km downstream of the confluence with the mine discharge water.
- Site 5: WR 2km DN - Wingecarribee River approximately 2km downstream of the confluence with the mine discharge water.
- Site 7: WR 3km DN - Wingecarribee River approximately 3km downstream of the confluence with the mine discharge water.

3.2.2.2 Discharge Monitoring Sites - Far

This site represents the Wingecarribee River at the edge of the mixing zone downstream of the adit discharge.

- Site 8: Biloela - Wingecarribee River at Biloela Camp Site approximately 6km downstream of the confluence with the mine discharge water.

3.2.2.3 Reference Sites

These reference sites indicate the ambient water quality within the Wingecarribee River immediately upstream of the adit discharge as well as within a tributary of the river within the same catchment.

- Site 2: WR Up - Wingecarribee River 100m upstream of the mine adit discharge.

- Site 6: Medway Rivulet approximately 100m upstream of the confluence with the Wingecarribee River.

3.2.3 Ambient Water Quality Monitoring Program

EPL 608 also specifies ambient water quality monitoring in the Wingecarribee River. The timing of the river monitoring generally corresponds to the discharge monitoring. These sites are historical sites which continue to be used to determine background water quality. The original four monitoring locations within the Wingecarribee River are listed below.

- Wingecarribee River upstream of the mine adit discharge at Old Hume Highway Crossing at Berrima (Licence Point 9).
- Wingecarribee River upstream of the mine adit discharge at Macarthur's Crossing (Licence Point 10).
- Wingecarribee River downstream of the mine adit discharge at Biloela Camp Site (Licence Point 11).
- Wingecarribee River downstream of mine adit discharge at Black Bob's confluence (Licence Point 12).

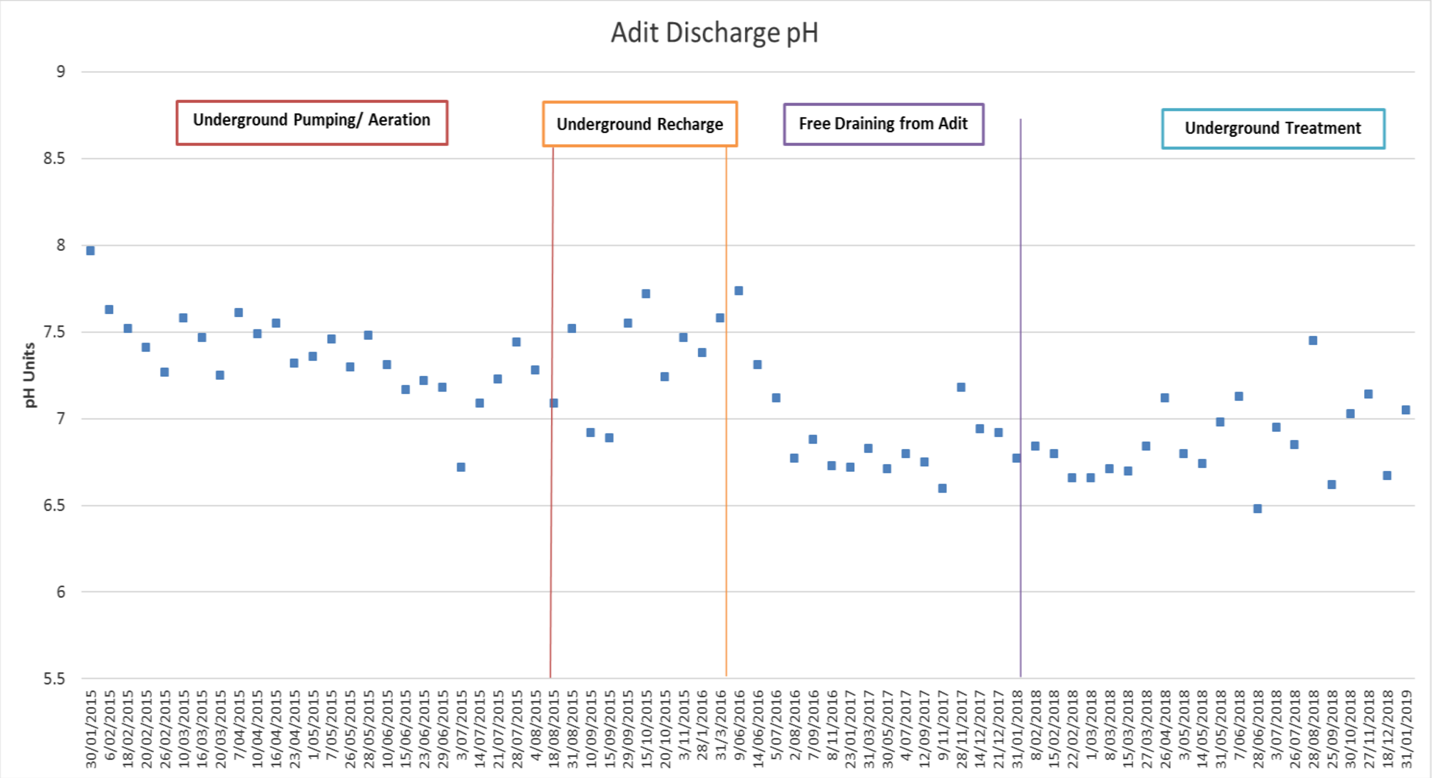
3.3 Results and Discussion

3.3.1 Adit Discharge

The following graphs provide a summary of discharge water quality prior to closure through to the underground treatment system. Noted on the graphs is the period immediately prior to the flooding of the mine, that is, when the original underground water management system was operating. The mine was allowed to flood in mid 2015 which involved the progressive removal of the internal pumping system. There was close to a 12 month period between the removal of the internal pumps and the commencement of free draining. This is the period when the mine was flooding up to the Pit Bottom Sump which overflowed into the Drain Adit and into the Wingecarribee River. The commencement of the new underground treatment system in early 2018 is also noted on the graphs.

In late 2018, the volume of water pumped from the mine increased in order to reduce the standing water level sufficiently to install a set of seven bulkheads. The purpose of the bulkheads is to determine the permeability of the overlying strata which will assist in determining potential final closure options. The increased pumping rate has reduced the treatment ability of the system which has influenced the water quality results. The influence to date has been small but detectable in the discharge results.

The results of the discharge water quality testing is shown on the following graphs.



Graph 3.1 – Adit Discharge pH

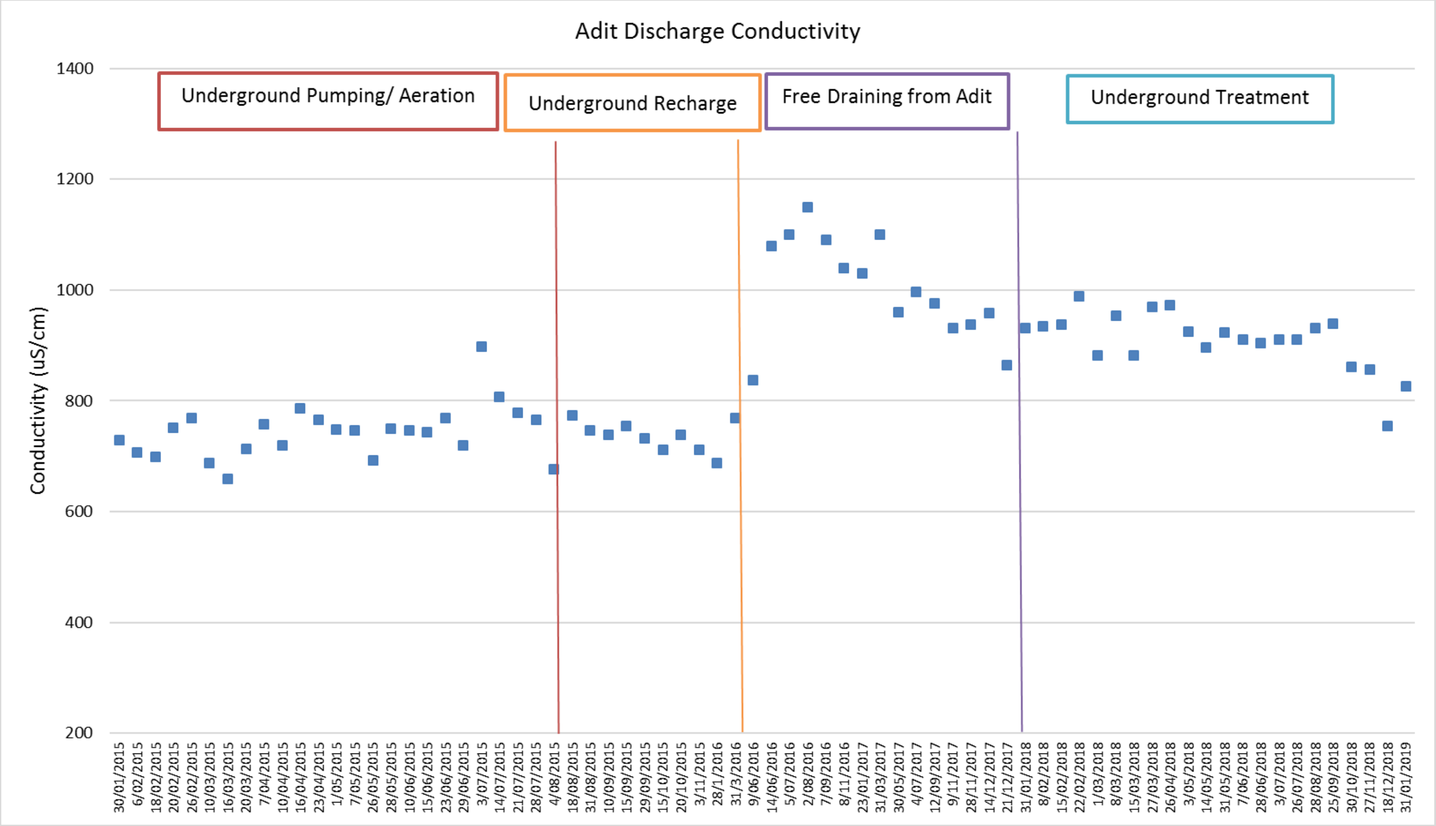
Although the pH increases during the treatment process, it drops prior to discharge. This is possibly a consequence of the water still needing to travel through the old workings after treatment in order to discharge at the licensed discharge point. As seen on Graph 3.1, there was a slight variation in discharge pH from mid-2018 as a result of the increased pumping. Although there has been a general increase in pH, it is still slightly below the level achieved while the mine was operating.

It should also be noted that when the mine was recharging (flooding) there was very little discharge from the mine occurring. What was discharged consisted of small amounts that were pumped from the pit bottom area, including the main Pit Bottom Sump in order to maintain access. Therefore, this water was already treated when the underground water management system was fully operational.

Over the past 6 months, the pH has shown greater variability than occurred while the mine was operating. This is possibly the result of a combination of the variability within the treatment system, the influences of water which may bypass the treatment system and the increased pumping rate in the latter part of the reporting period.

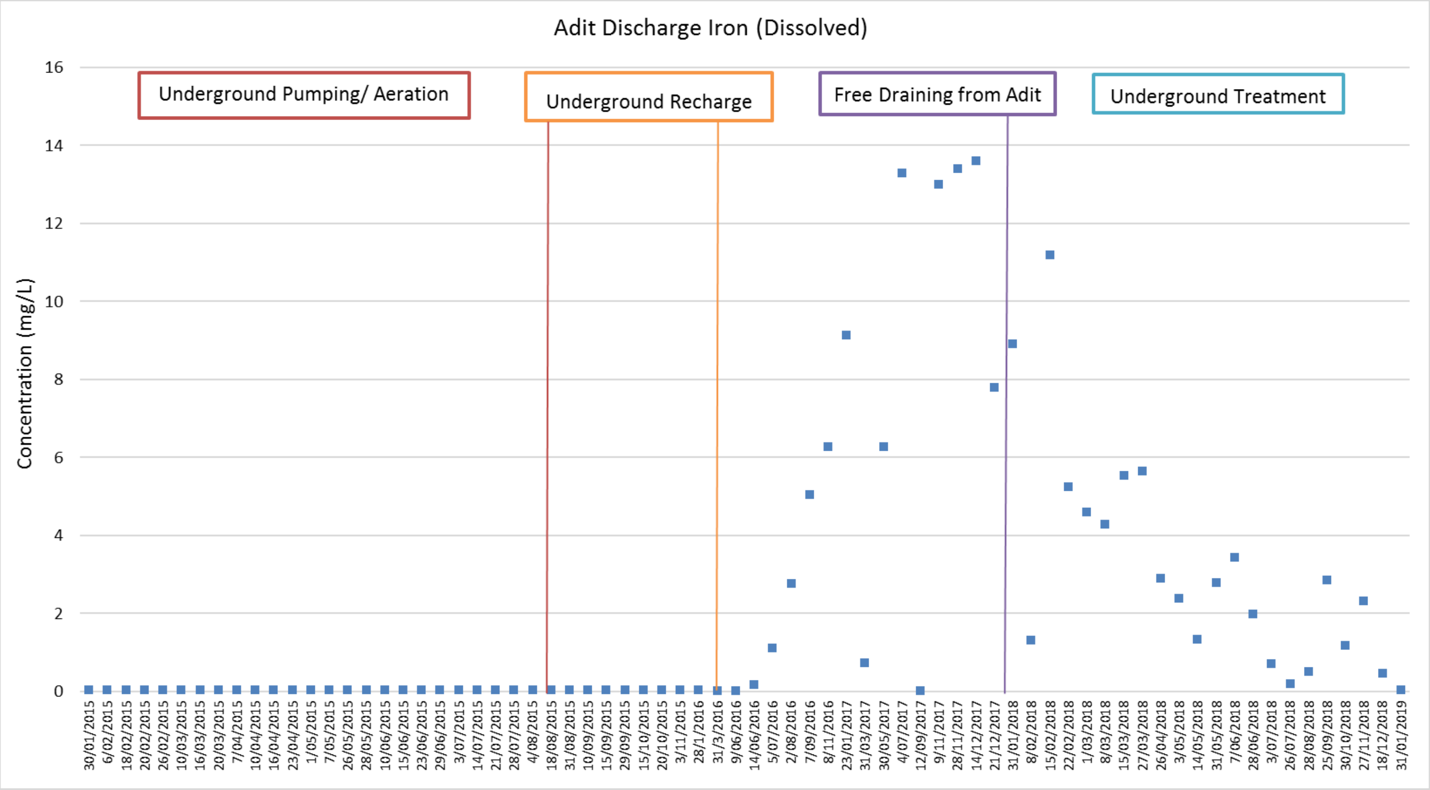
Graph 3.2 below shows the results for conductivity. This is generally a measure of salt but also other dissolved minerals. The cause of the initial increase in conductivity followed by a decrease while the mine was free draining is unknown but is likely a function of the change in dissolved solids loading of the water.

Although commonly linked with salt content, conductivity can also measure other dissolved solids such as Iron and Manganese. As such, any change in dissolved iron and manganese can also influence conductivity levels. Towards the end of this reporting period, conductivity levels approached the historic long-term average discharge from the mine.



Graph 3.2 – Adit Discharge Conductivity

Although historically the mine discharged water with a conductivity below 900 μS/cm, it had on many occasions discharged water up to 1,000 μS/cm. This fluctuation in conductivity was considered to be caused by local geological conditions but also groundwater recharge could also be a factor (Berrima Colliery Water Management Plan 2013).

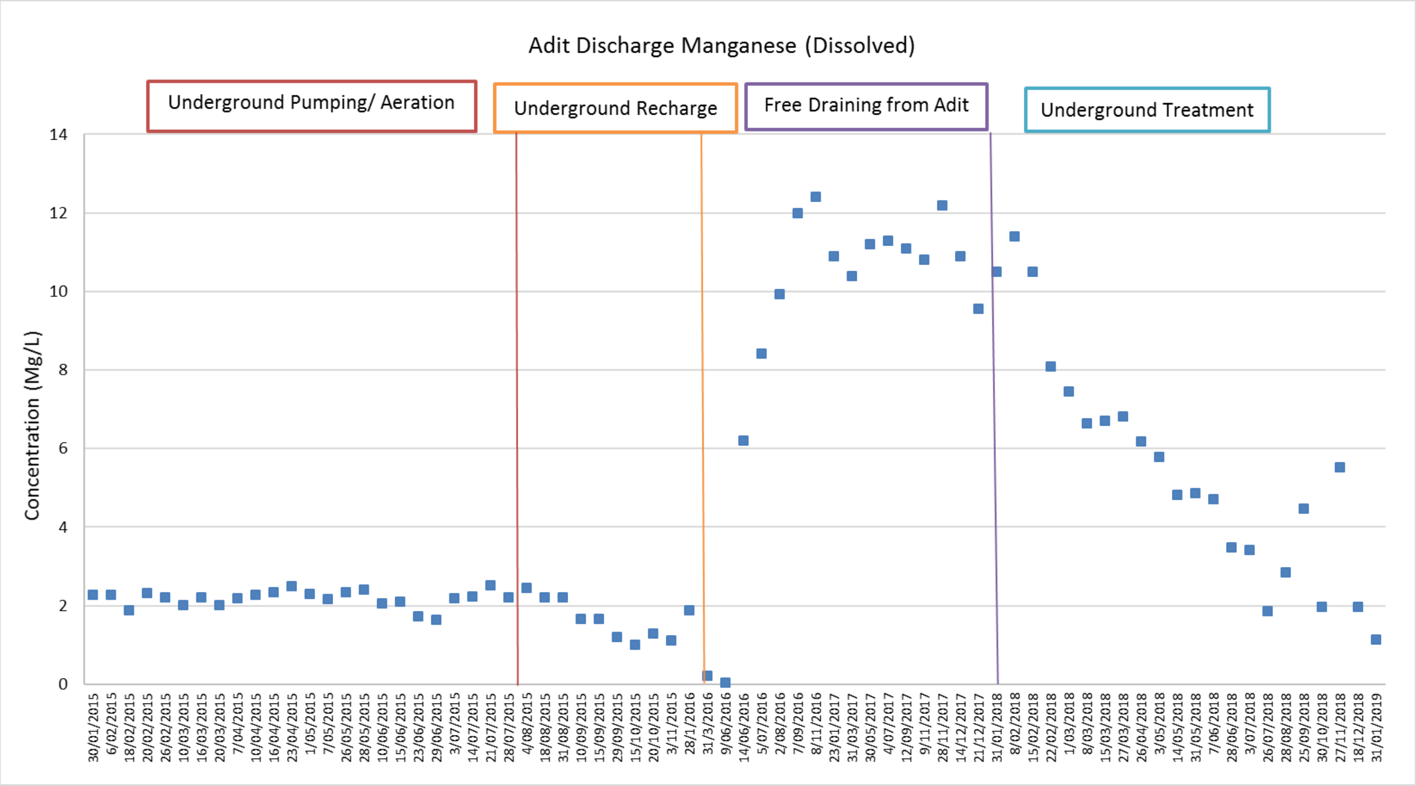


Graph 3.3 - Adit Discharge Iron Concentration

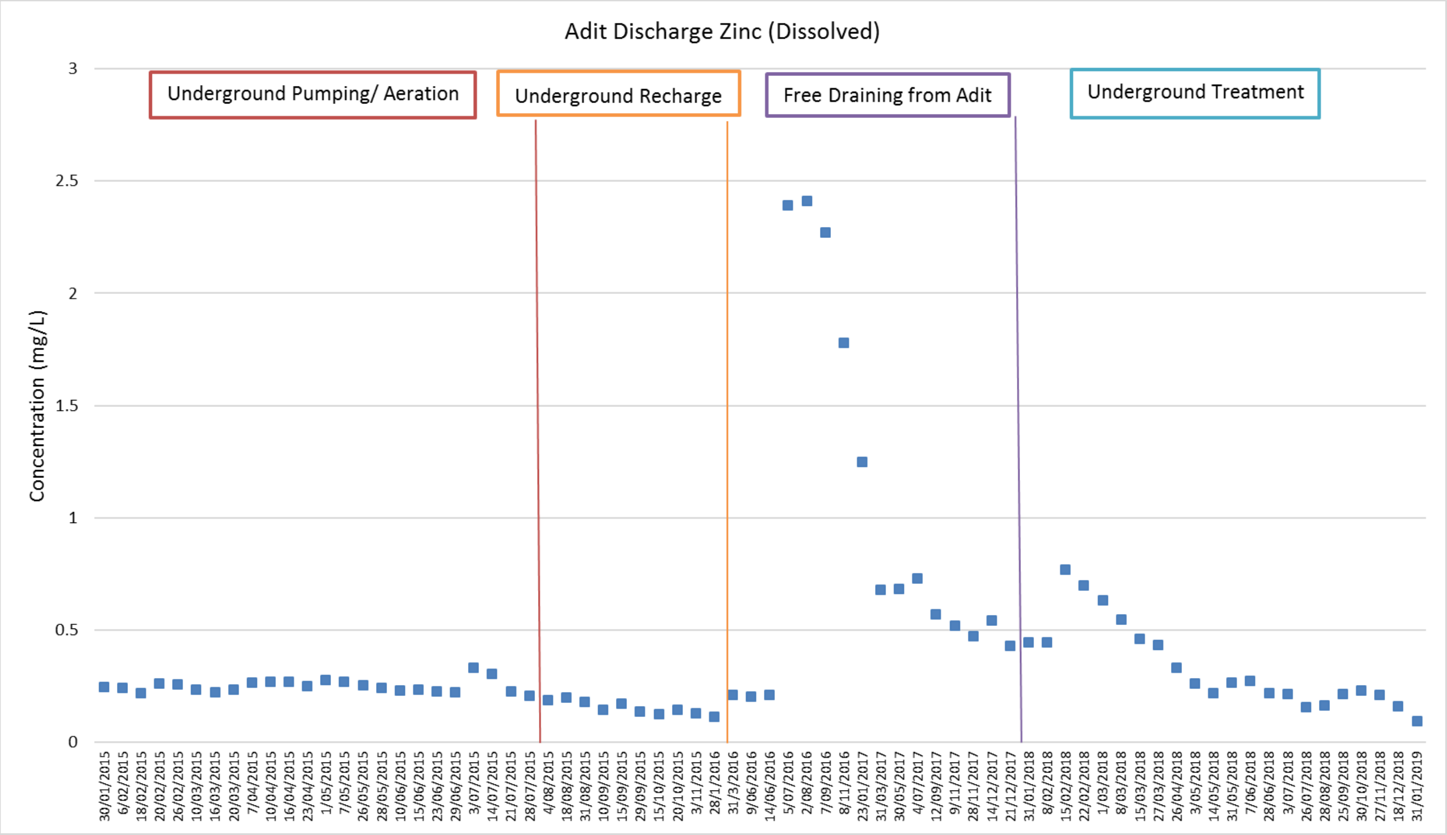
Graph 3.3 shows that the concentration of Iron prior to the mine free draining was very low. Once this previously treated water was removed during the free draining period, the concentration of Iron rapidly increased. Once the new treatment system was installed, the Iron concentration progressively reduced to near historic minimal concentrations. There was a slight increase in Iron following the increase in pumping rate but the magnitude of the change was small.

Manganese, Zinc and Nickel follow a similar trend of initial deterioration during free draining followed by progressive improvement following implementation of the new treatment system. This is shown on Graphs 3.4 to 3.6. Despite these minerals being much harder to remove using a passive treatment system, the results are encouraging with the most recent results being very similar to historic levels.

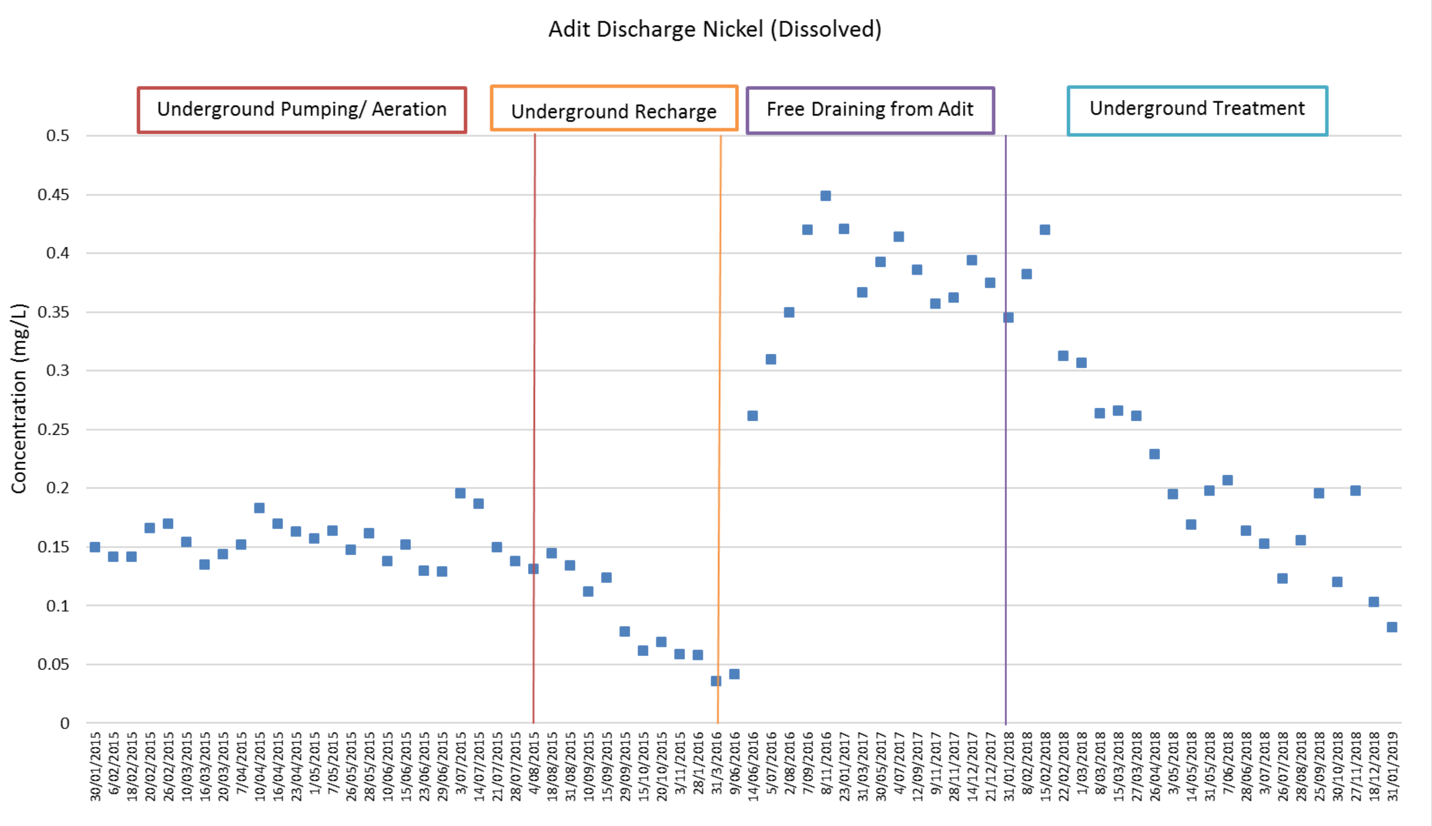
There was a marginal increase in metal concentration during the increased pumping undertaken in late 2018, however the most recent results in January 2019 were very close to the historic levels for Iron, Manganese, Zinc and Nickel.



Graph 3.4 – Adit Discharge Manganese Concentration



Graph 3.5 – Adit Discharge Zinc



Graph 3.6 – Adit Discharge Nickel

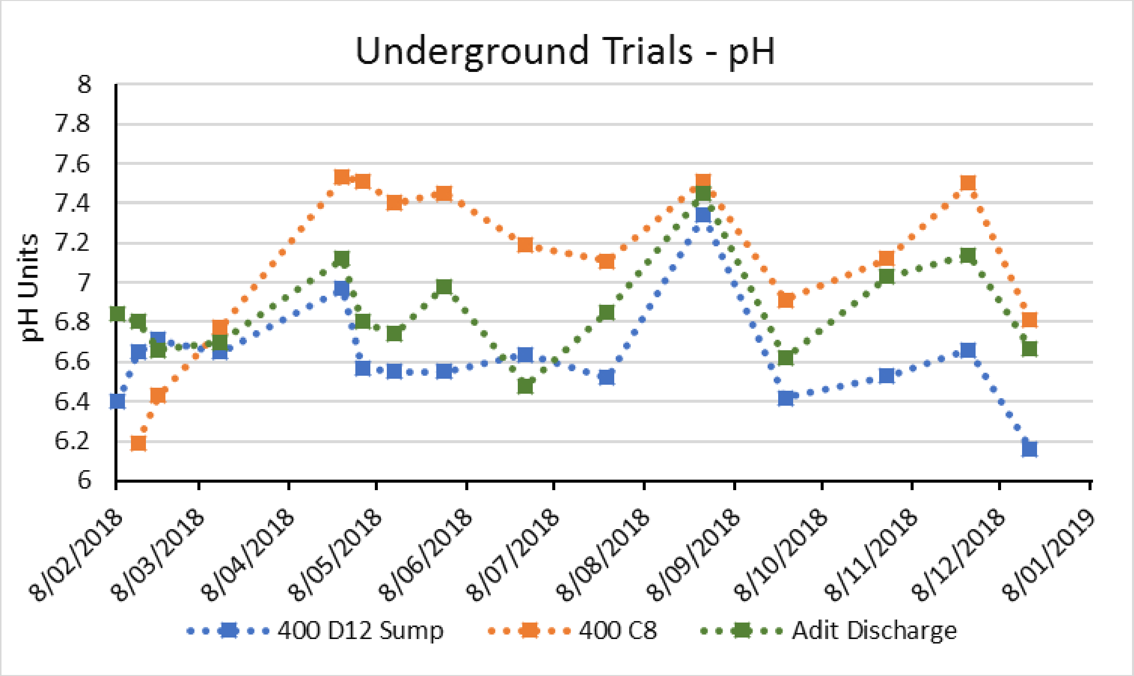
The results of the discharge monitoring demonstrate that water quality leaving the mine has significantly improved since the introduction of the underground treatment system. The discharge quality has now approached, and in some instances better than the long term average discharge from the mine. The results of the underground treatment system is provided in the following sections.

3.3.2 Underground Treatment System

As a result of poor water quality discharged from the mine, a Pollution Reduction Program was conditioned on EPL608. Conditions E2.4 and E2.5 required Boral to carry out water treatment trials. The aim of the program is to reduce the mineral content in the discharge water. Iron is the main mineral that caused the discoloured discharge into the Wingecarribee River, however the concentration of other minerals also need to be reduced to improve overall water quality. The initial pit top water treatment trials involved constructing a series of limestone channels to run tests on aeration and pH adjustment on a component of mine water delivered to the surface via the existing water supply line which runs up the drift.

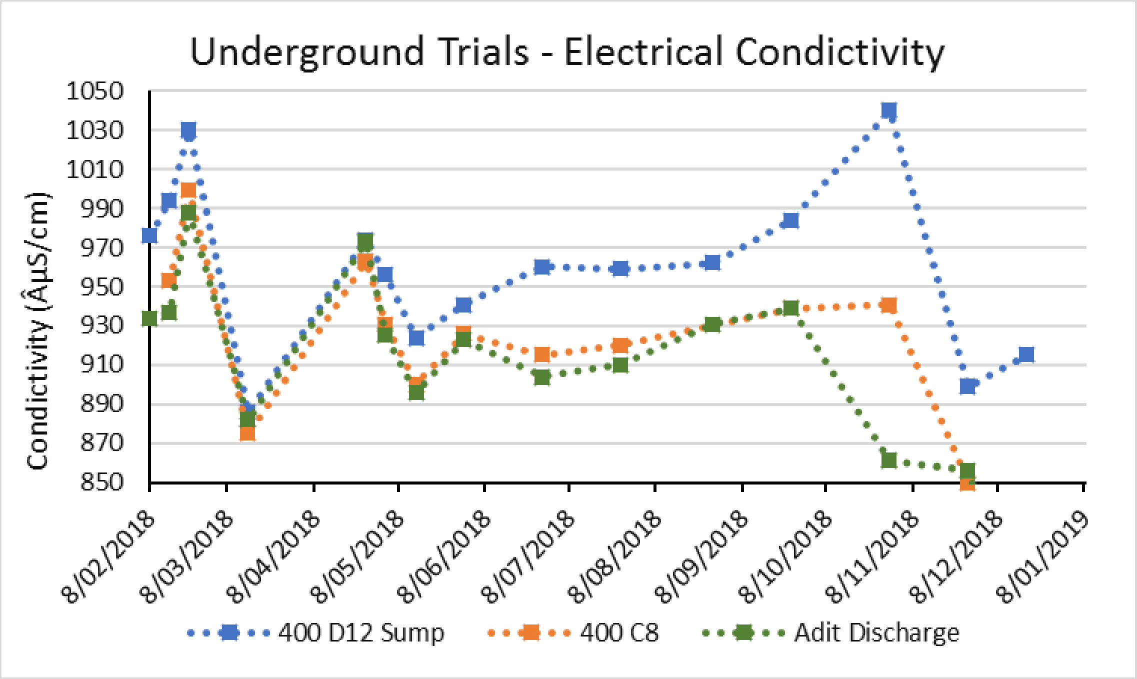
The surface trials were successful with the removal of most of the Iron and approximately 25% of dissolved Manganese. The system was then extended to the underground workings in January 2018. This included the installation of a separate pump line from the flooded section of the mine workings which discharged into a previously dry roadway within the old workings. The roadway was lined with limestone aggregate while aeration was provided by an intervening above ground metal sump. Venturis were installed in the pump line however these did not prove successful as adequate aeration occurred during the pumping process.

Graphs 3.7 to 3.15 show the results of the underground water treatment system. Three monitoring sites are shown. The D12 Sump which is at the commencement of the treatment system, C8 site which is at the return of the treatment system and the Adit Discharge which is after final settlement in the Pit Bottom Sump.



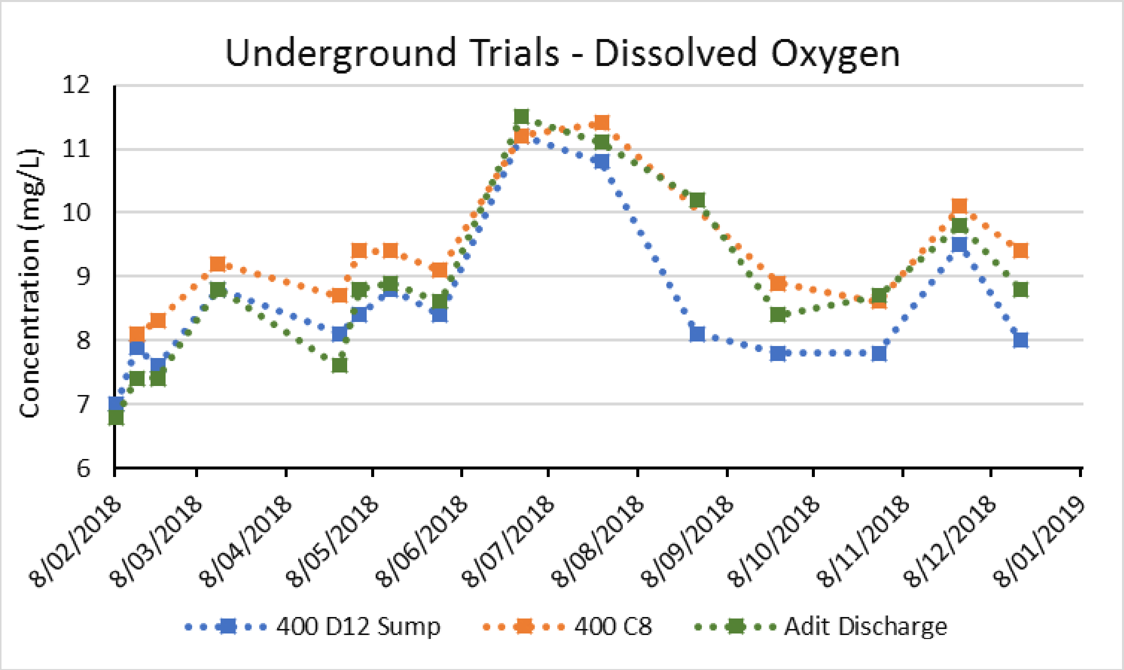
Graph 3.7 – Treatment System pH

The results for pH demonstrate that the treatment system is functioning largely as designed. The pH rises almost one unit following treatment although a portion of this alkalinity is lost prior to discharge. The overall pH level at the end of the treatment system is within the range of historic discharge levels however there is a drop between the end of the treatment system and the discharge point.



Graph 3.8 – Treatment System Electrical Conductivity

As observed above in Graph 3.8, Electrical Conductivity tends to follow a consistent trend throughout the treatment system. Salinity levels are generally higher within the flooded mine section (400 D12 Sump) than treated water at 400 C8 and at the Adit Discharge. This is considered to reflect the removal of metals and Sulphate however there is also a comparitive trend ocuring as well, that is, higher initial conductivity results in higher discharge conductivity and vice versa.

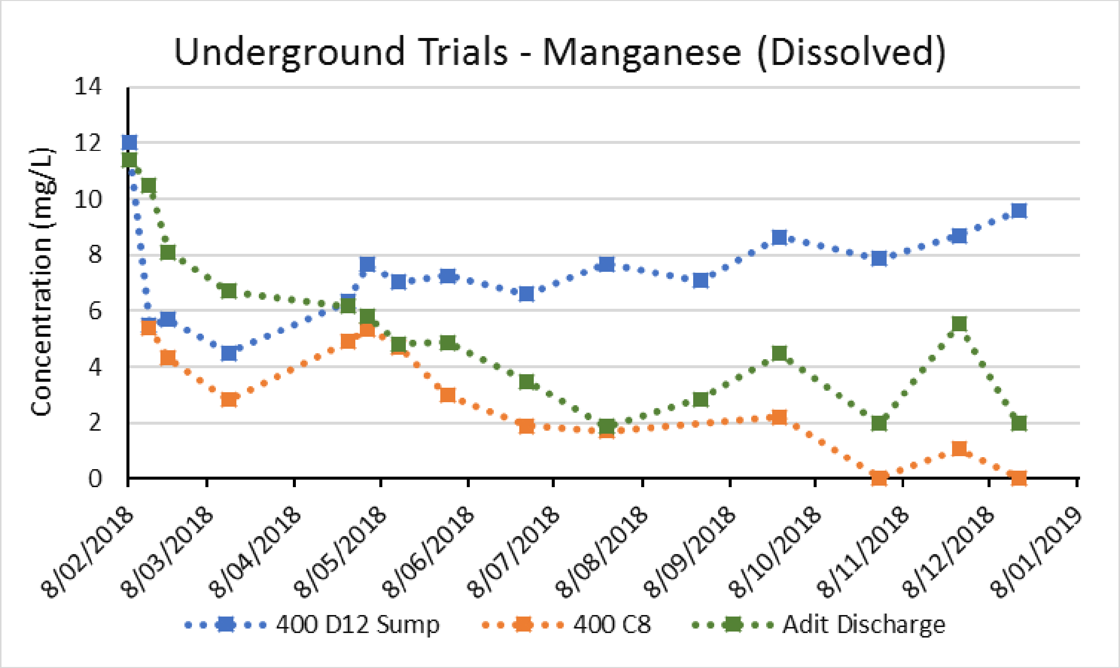


Graph 3.9 – Treatment System Dissolved Oxygen

The Dissolved Oxygen has shown a variable trend since monitoring of the treatment system began as seen in Graph 3.9 above. Despite the variability over time, all samples sites follow a similar trend with the treated water consistently having a higher concentration than the untreated water. This indicates that the oxygenation process at the commencement of the treatment system is effective. Dissolved Oxygen however has fallen with the increased pumping rate which could also partly account for the increase in metal content at the discharge point in late 2018.

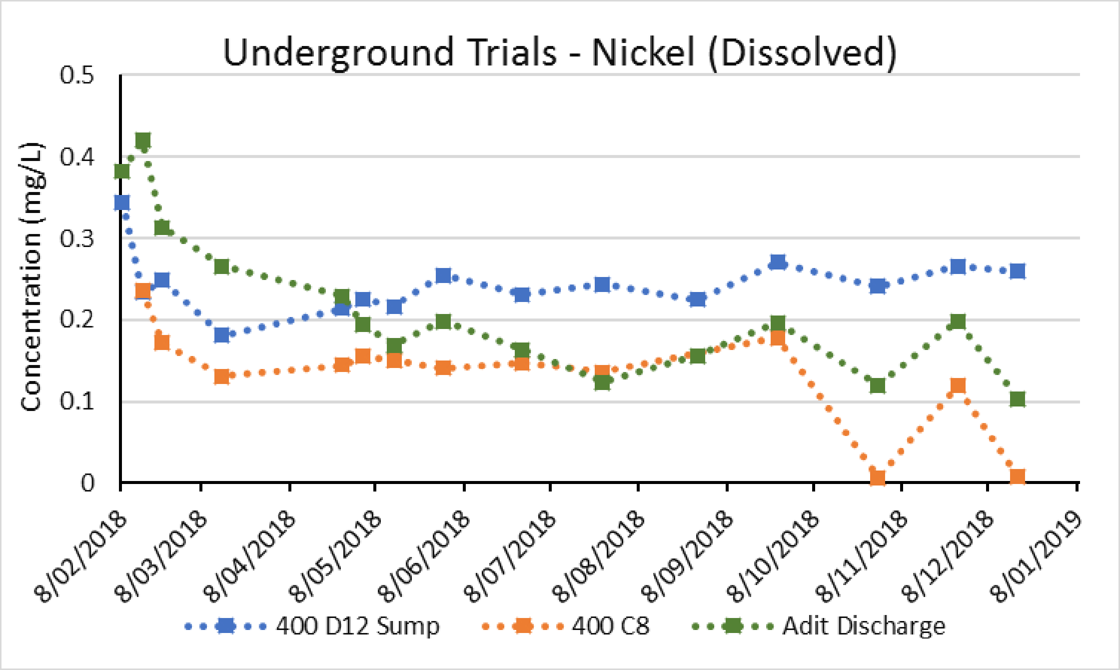
Changes were made to the underground water management system during the 4th Quarter of 2018. This involved pumping additional water from the mine in order to lower the flooded mine workings to enable access to construct additional bulkheads.

The results for component metals is provided in Graphs 3.10 to 3.14. The results show the general improvement over time within the treatment system.

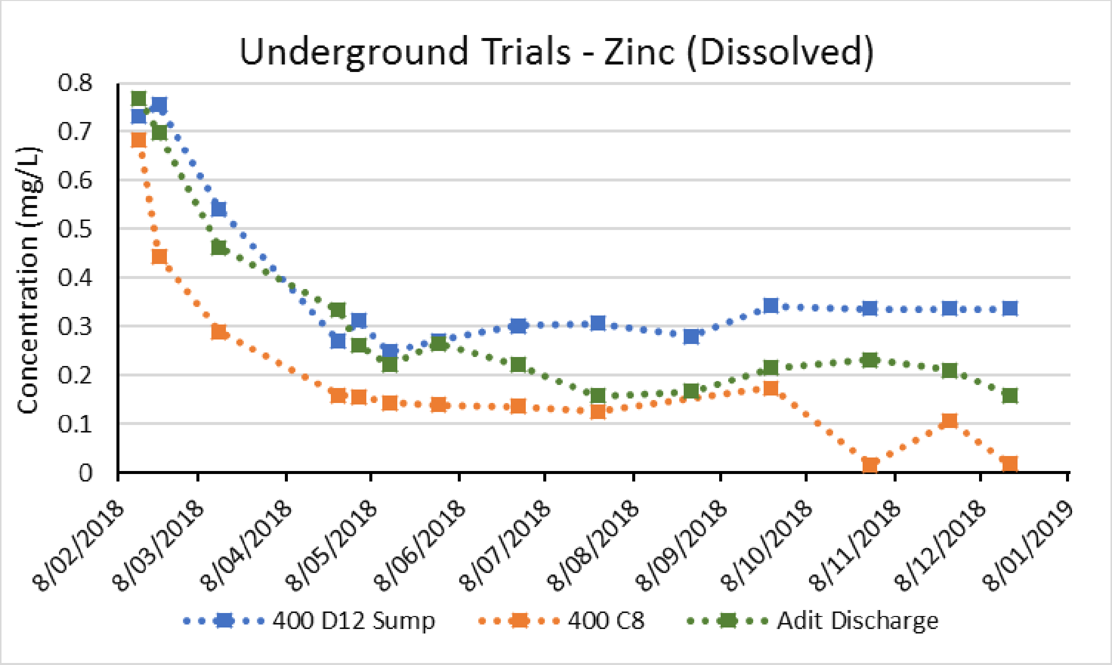


Graph 3.10 – Treatment System Manganese (Dissolved)

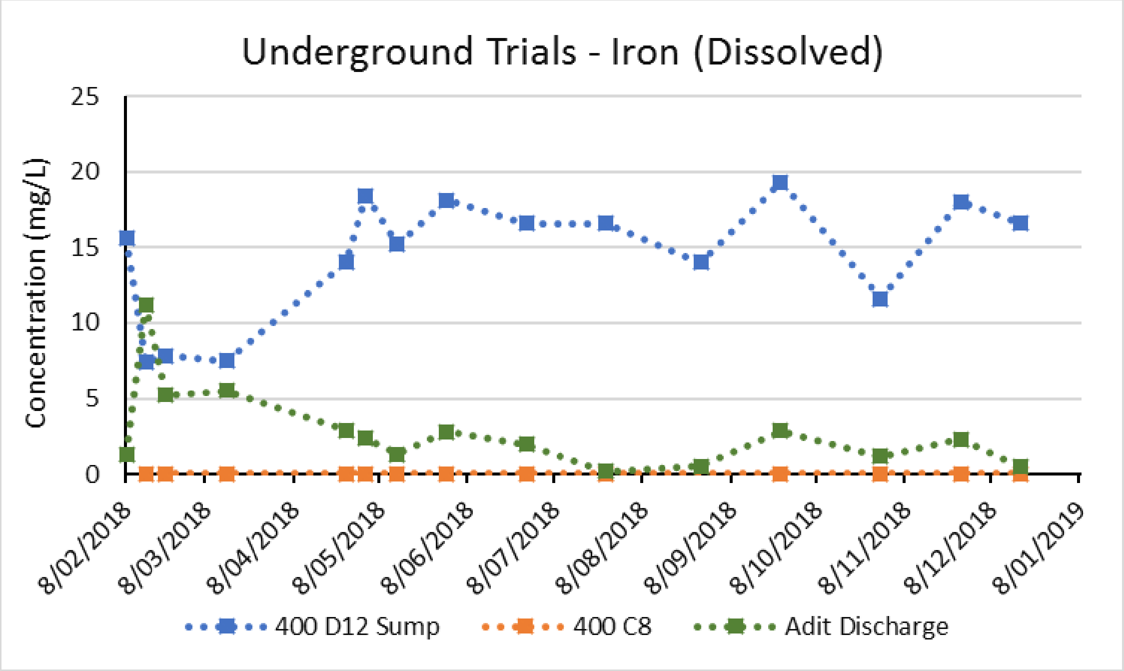
Based on the initial water modelling undertaken on the surface trials, Manganese should be precipitated as Birnessite which is an oxide mineral of manganese along with calcium, potassium and sodium $(Mn^{4+}, Mn^{3+})_2O_4$ 1.5 H_2O. Iron precipitates would be in the form of Ferrihydrite, $Fe^{3+}O_2$ 0.5(H_2O).



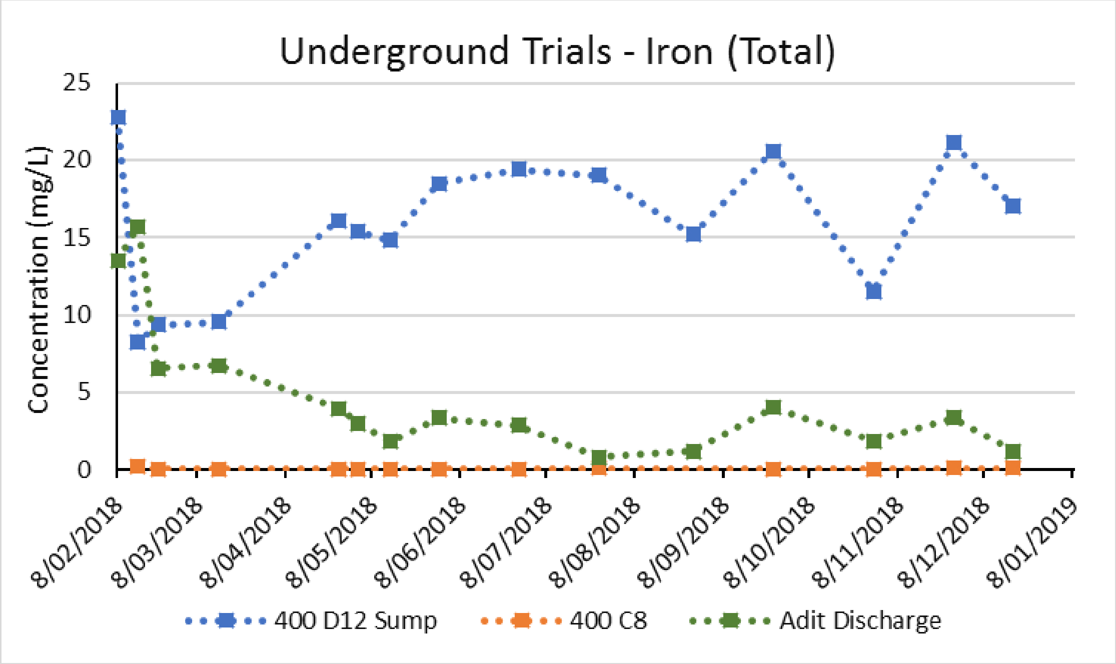
Graph 3.11 – Treatment System Nickel (Dissolved)



Graph 3.12 – Treatment System Zinc (Dissolved)



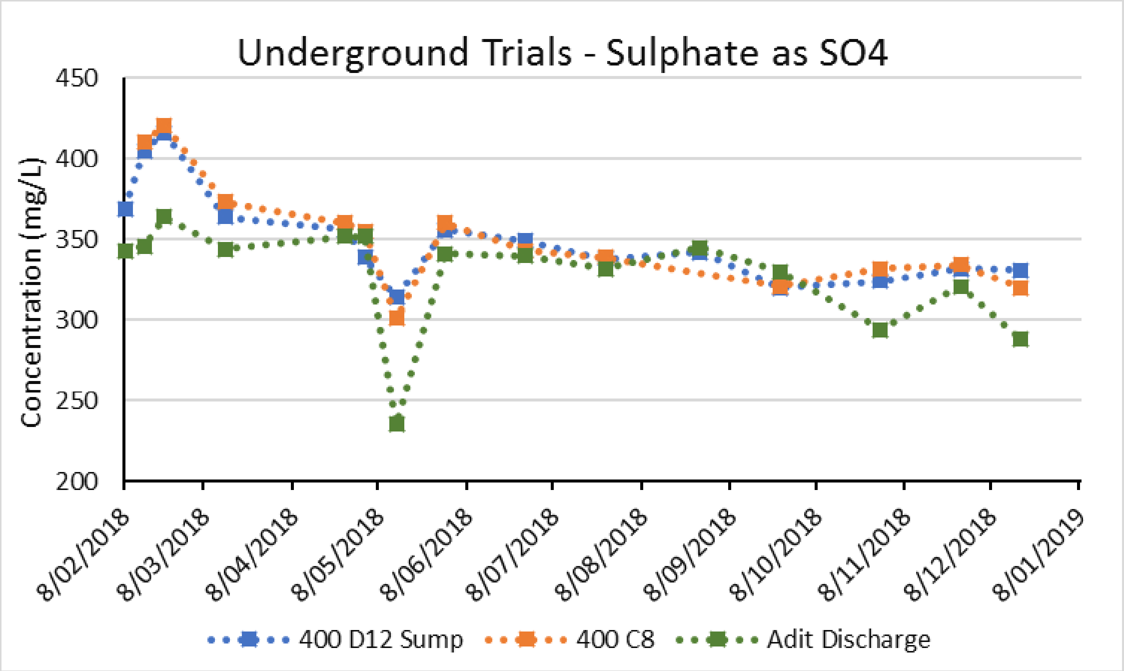
Graph 3.13 – Treatment System Iron (Dissolved)



Graph 3.14 – Treatment System Iron (Total)

The greatest improvement in water quality has occurred with Iron. Graphs 3.13 and 3.14 show that at the end of the treatment system (400 C8) nearly all detectable Iron has been removed. These graphs also show a reduction in both total and dissolved Iron concentrations at the discharge point compared to before treatment began. The increase in Iron from September until the end of 2018 has been caused by the increased pumping rate which necessarily involves some bypassing of the treatment system.

A similar trend - although not as pronounced - has occurred with Manganese, Nickel and Zinc in Graphs 3.10, 3.11 and 3.12 respectively. The overall concentrations of these metals at the discharge point is close to the historic averages and a significant improvement on the levels which occurred during the initial phase of free draining. These minerals are more difficult to remove via a passive treatment system using only aeration, limestone pH adjustment and settlement such as is being currently used. Concentrations could be further reduced with artificial pH adjustment using chemical alkali however this is not considered desirable nor necessary at this stage.



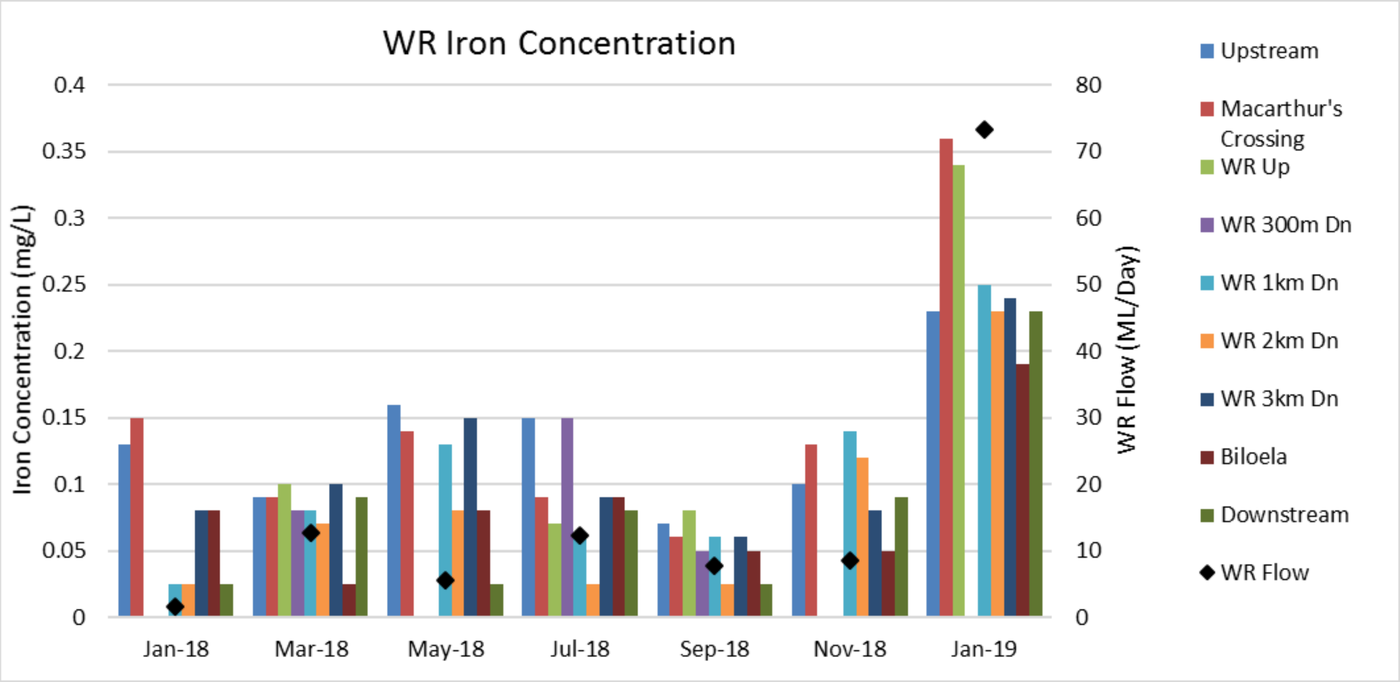
Graph 3.15 – Treatment System Sulphate

The concentration of Sulphate, which is also a component of the conductivity level was analysed as shown on Graph 3.15. The treatment system using limestone binds Sulphate with Calcium to form gypsum which is insoluble. The gypsum is formed initially as a coating on the limestone aggregate but is often washed off and is then collected in the sumps as a sediment. The removal of approximately 9% of the free sulphate represents the largest proportion of the conductivity reduction achieved by the treatment system. The remaining reduction would be attributed to the reduction in mineral content. Although there has been a slight decrease in the concentration of Sulphate at the discharge point, the reduction is too small to attribute this to any specific activity underground.

3.3.3 Wingecarribee River Water Quality Monitoring Results

The raw monitoring results are provided in Appendix C while a summary of the average results is provided in Graphs 3.16 to 3.20. The graphs show the individual monthly results along the Wingecarribee River grouped and colour coded according to sample sites. Samples from Site 2: WR Up and Site 3: WR 300m Down were sampled only in the months March, July and September due to access difficulties. An alternative WR Up site at the Colliery's Drift was sampled in January 2019, in accordance with the EPL. Also shown on the graphs is the corresponding monthly river flow. This is important to show on each graph as it provides some context in relation to water quality results.

For each of these graphs, the mine discharge occurs between the results for WR Up and WR300m Down. The diamonds on each set of monthly bar graphs shows the monthly water flow. Comparing the results of river water quality and the discharge water quality (Graphs 3.1 to 3.6) provide useful information on both the river health and influence that the mine has on receiving water quality.

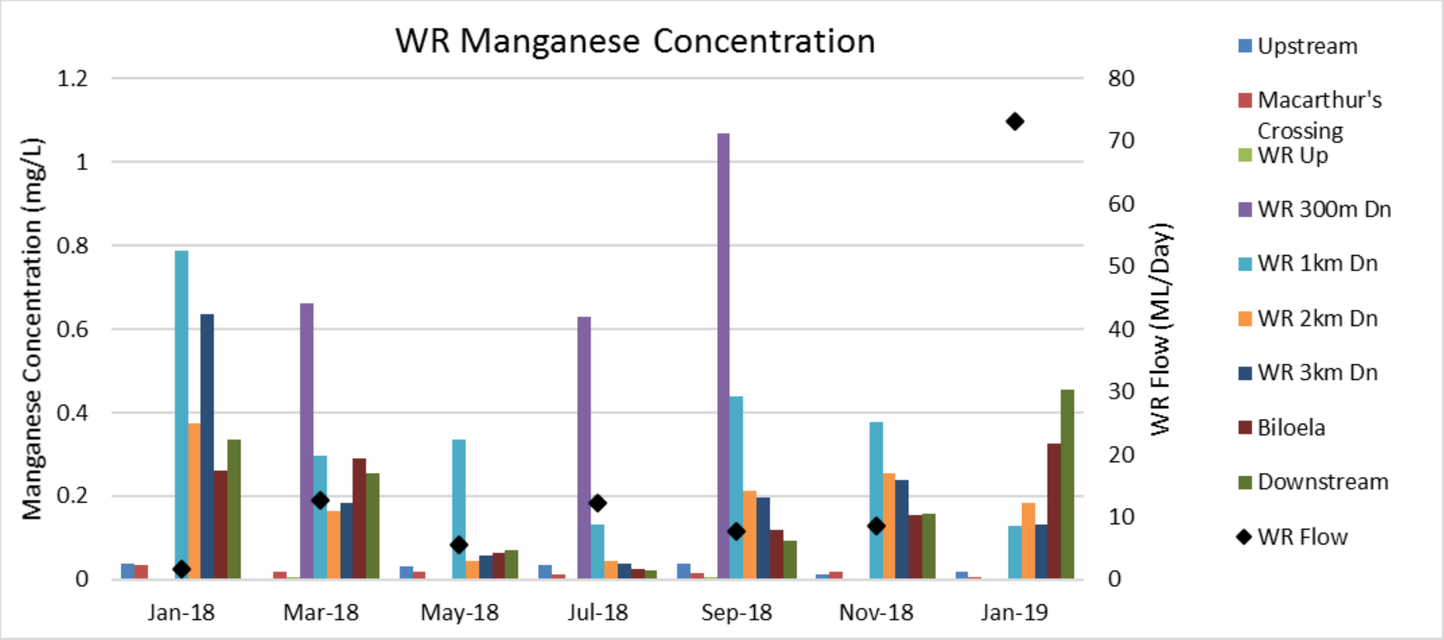


Graph 3.16 – Wingecarribee River Iron Concentration

In low flow, the concentration of Iron is generally low. There is an influence each month downstream of the mine discharge however this influence is minor compared to river flow. When the mine was discharging elevated Iron concentrations early in 2018, there were higher concentrations of Iron in the river upstream of the discharge. In high river flow periods such as January 2019, the concentration of Iron was elevated both upstream and downstream of the mine discharge. During this month, the mine discharged Iron at levels less than laboratory detection limits.

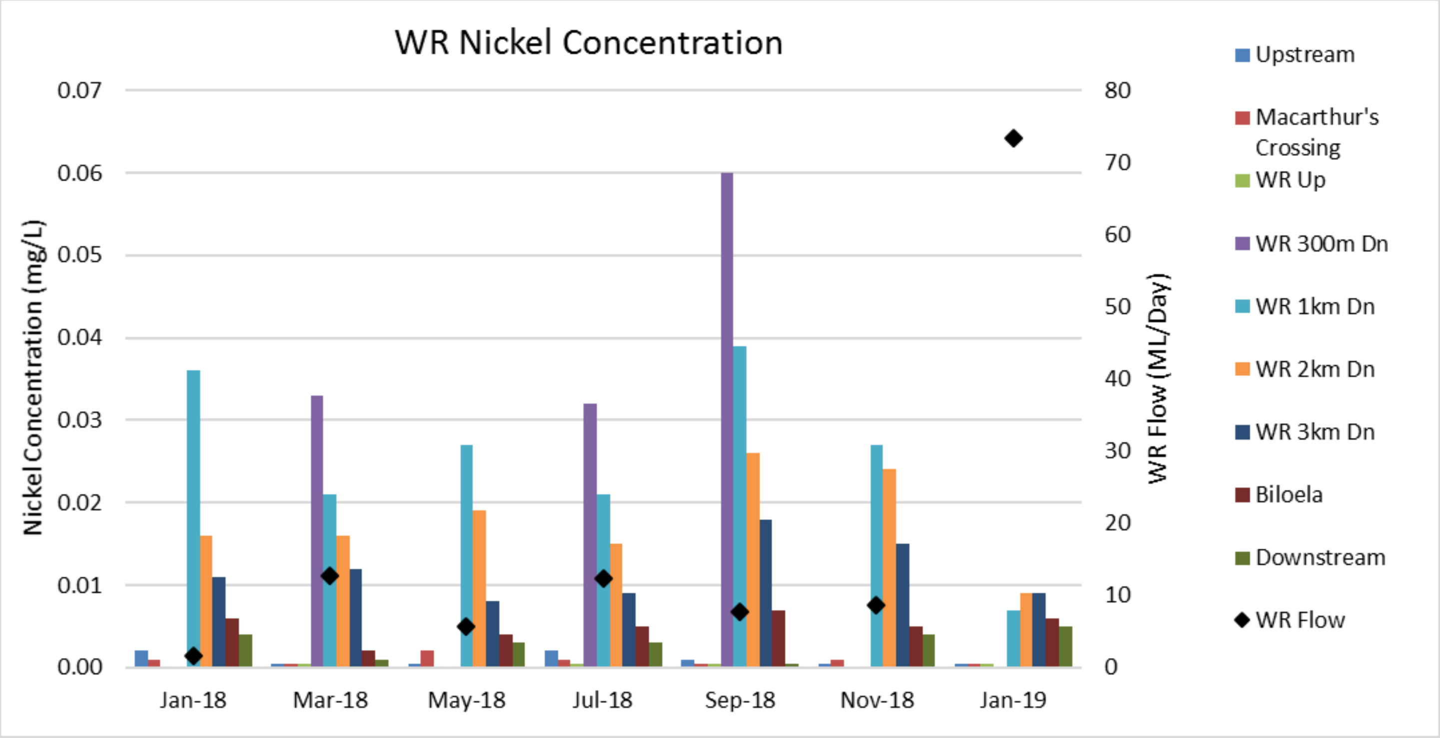
Similar results occurred mid-year although not as pronounced. Iron concentrations tend to be higher when the river is flowing and lower when the river flow is very low.

The opposite seems to occur with Manganese, although there is also greater variability and the influence of the mine discharge is more pronounced. There is generally a noticeable increase in Manganese downstream of the mine discharge. The increase is less pronounced in periods of high flow.



Graph 3.17 – Wingecarribee River Manganese Concentration

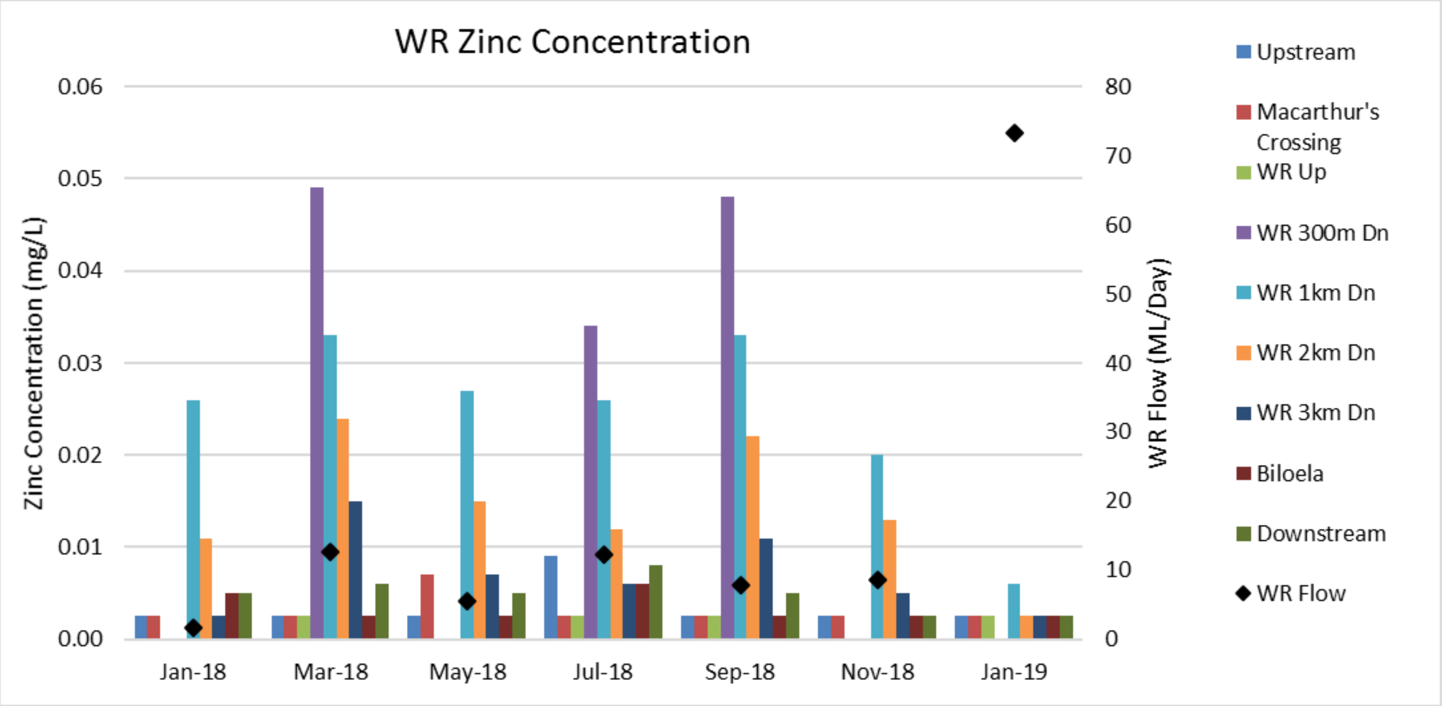
The average Manganese concentration shown on Graph 3.17 decreases rapidly through the mixing zone but remains slightly higher further downstream than upstream. Manganese concentrations downstream from the adit discharge range from 0.13 mg/L to 0.79 mg/L. This is similar to the historical situation and concentrations downstream are all below the 95% ANZECC default criteria of 1.9 mg/L.



Graph 3.18 – Wingecarribee River Nickel Concentration

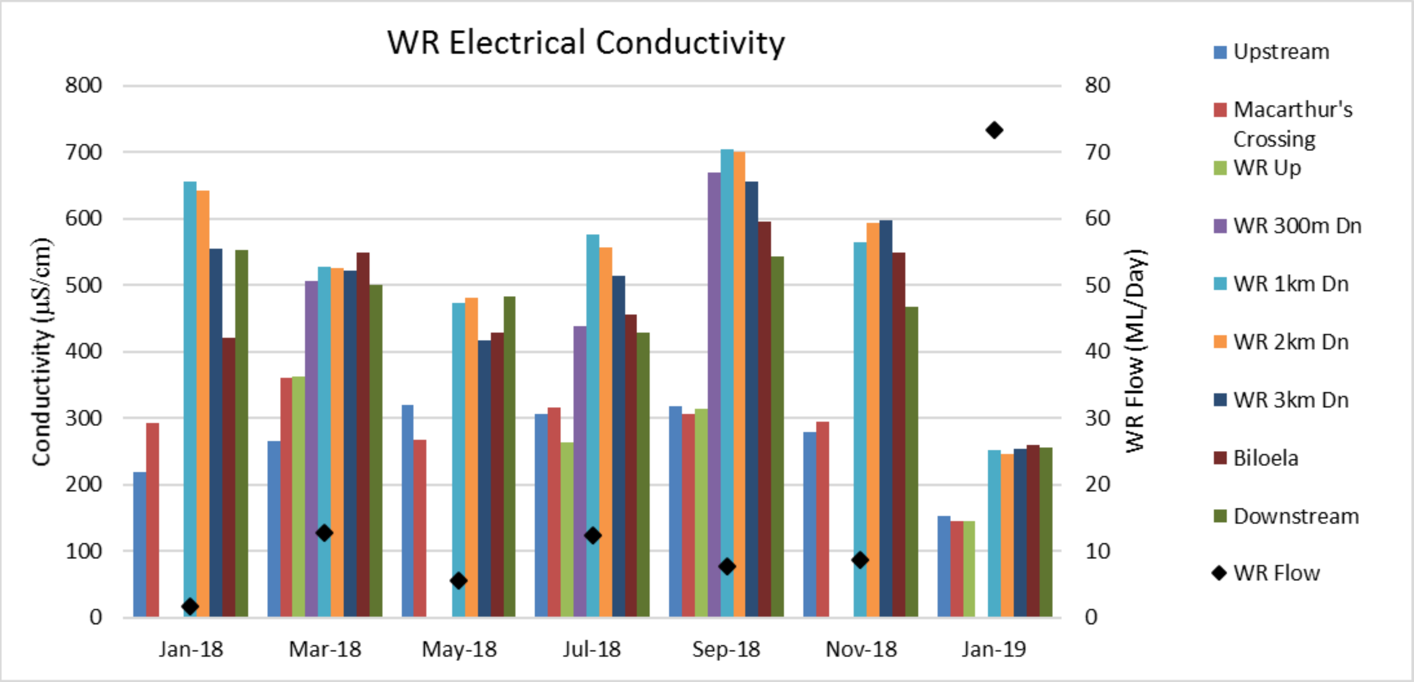
As with Manganese, Nickel concentrations in the river tend to follow river flow but with a greater influence registered from the mine discharge. The concentration of Nickel below the discharge point peaked in September 2018. This month corresponded to a relatively low concentration but an increased mine discharge volume at a time when the river flow was low.

The concentrations of Nickel and Zinc downstream rapidly reduce and fall below the ANZECC 2000 95% ecosystem protection default guideline values of 0.011mg/L and 0.008 mg/L respectively at the Biloela site.



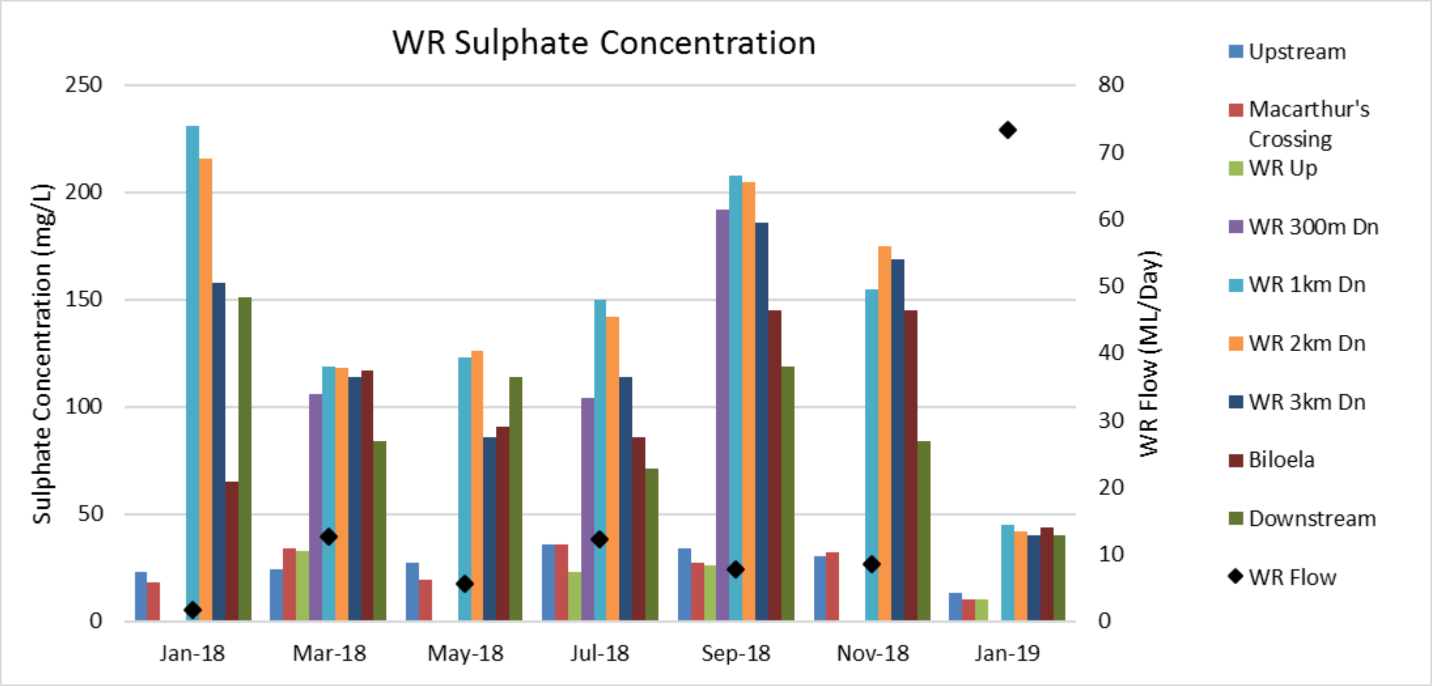
Graph 3.19 - Wingecarribee River Zinc Concentration

As shown in Appendix C, the pH of the mine discharge is currently averaging 0.5 pH units below its long term average. The lower pH at the Adit Discharge does not appear to be influencing downstream water quality which ranged from 7.49 to 7.56 pH Units.



Graph 3.20 – Wingecarribee River Average Electrical Conductivity

The higher conductivity of the discharge causes an increase in conductivity within and beyond the mixing zone, as seen above in Graph 3.20. The increase is approximately 200 µS/cm. The conductivity is generally below 500 µS/cm which is considered fresh and non-saline.

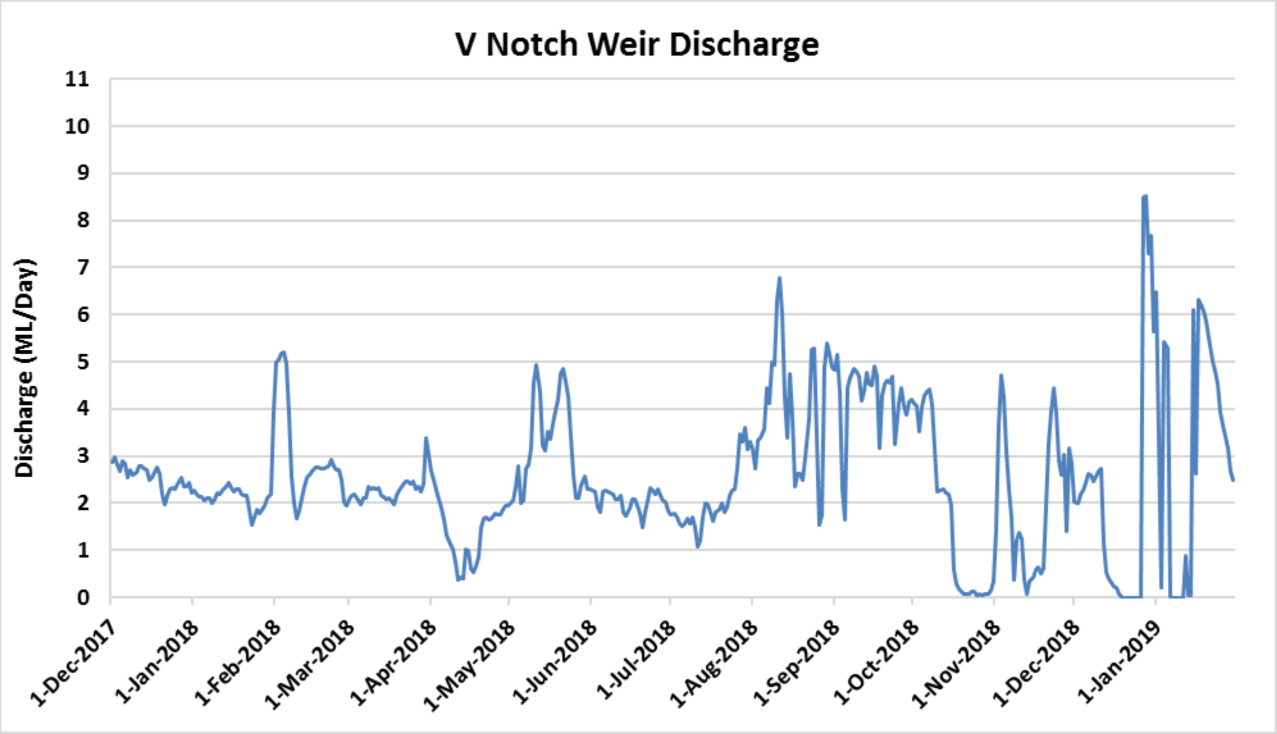


Graph 3.21 – Wingecarribee River Average Sulphate

The concentration of sulphate shown on Graph 3.21 remains relatively constant downstream of the discharge point. There is little fluctuation in levels despite increasing distance. The measured levels are generally around levels of 100 mg/L which is considered within the normal environmental range.

3.3.4 Discharge Volumes

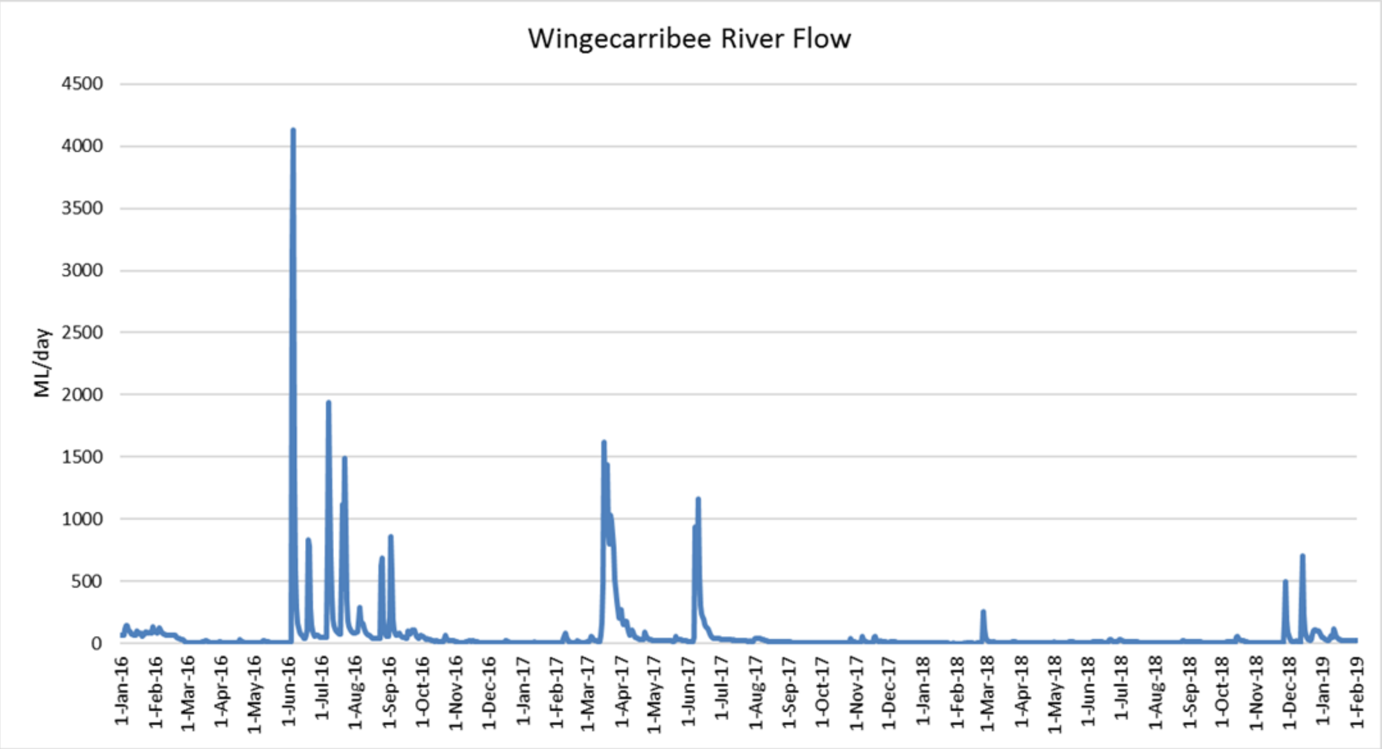
During 2018, the mine discharged an average of 2.5 ML/day which is below the long term average of 2.7 ML/day. The daily discharge data is provided below in Graph 3.22.



Graph 3.22 – V Notch Weir Discharge

3.3.5 Weather Data and River Flow

Over the past 12 months, most of NSW has been in drought conditions. Other than three storm events on the 26th February, 29th November and 14th December 2018, rainfall has been very low. These events resulted in 256 mm, 494 mm and 705 mm of rainfall respectively, which saw a corresponding increase in river flow. However as shown in Graph 3.23 below, there has been very little flow in the river since July 2017.



Graph 3.23 – Wingecarribee River Flow

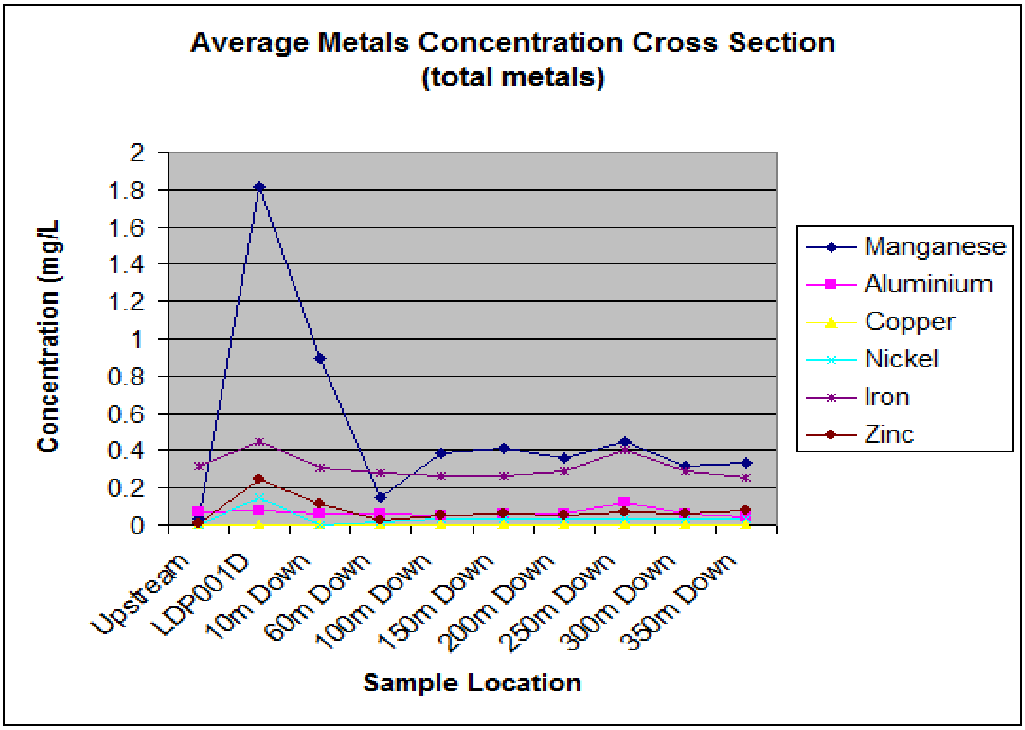
The consequence of low flow conditions is that the discharge from Berrima Colliery becomes the dominant water source for the river. Mixing becomes non-existent in very low flow conditions and other natural groundwater baseflow contributions can influence overall water quality.

Despite some rainfall occurring and recent water transfers by WaterNSW there has been no peak flows which would result in flushing of the river. The high flow events, which see over 1,000 ML/day flows are essential to maintain a healthy river.

3.4 Summary of Mixing Zone Water Quality 2012

Graph 3.24 shows a summary of water quality data obtained from the mixing zone in 2012. The results are a summary of the data provided in the Berrima Colliery Water Management Plan 2012. The monitoring sites were slightly different in 2012 and included sites closer to the adit discharge but also include the 300m downstream site as well as data from Biloela.

This data was taken at a time when the mine was fully operational, and the discharge quality was typical of the long term averages. The flow in the Wingecarribee River at the time these samples were taken were also low and between flushing events.



Graph 3.24 – Wingecarribee River Quality Mixing Zone 2012

A comparison of 2012 data with 2018 data is provided in the following table.

Table 3.1 – Water Quality Comparison 2012 With 2018

Parameter (mg/L)	Average Concentration 2012 (300m downstream)	Average Concentration 2018 (300 m downstream)
Manganese	0.32	0.87
Iron	0.29	0.76
Nickel	0.0345	0.0325
Zinc	0.465	0.057
	Downstream Concentration 2012	**Downstream Concentration 2018**
Manganese	0.076	0.23
Iron	0.48	0.27
Nickel	0.0049	0.0036
Zinc	0.01	0.006

From the above data, it can be seen that the concentration of Iron and Manganese 300 m below the discharge point is higher in 2018 compared with 2012, however the concentration of Nickel and Zinc are the same. Further downstream it is only Manganese that is higher in 2018 compared with 2012.

3.5 ANZECC 2000 Assessment

Table 3.2 shows the established SSTV for the receiving waters of the Wingecarribee River. This has been established using the 80th percentile Macarthur's Crossing results taken over 24 months up until 2013. The edge of the mixing zone is taken as the Biloela Camp Site downstream of the mine discharge. This site is also specified on the Performance Monitoring Program objectives as needing to meet the trigger values for primary industries and recreational water quality and aesthetics. The 2013 data represents the last period when the mine was operational which is appropriate to use given the change in discharge quality and the need to implement further underground treatment.

Table 3.9 – Edge of Mixing Zone ANZECC Assessment

Parameter	**Upstream Site (80th percentile)**	**Downstream Site (80th percentile)**	**ANZECC 95% Default / EPA**	**Recreation guidelines ANZECC**	**Mixing Zone 2013**	**Biloela SSTV**	**Biloela 2018 Average**
pH	7.8	7.49	6.5 to 8.5	6.5 to 8.5	7.5	7.5	7.5
EC (uS/cm)	279.5	312.5	350	1,500 *	430	350	460
TSS	17	16.5	50	<20% of natural	12	12	4.2
Total Nitrogen (mg/L)	0.335	0.30	0.25	10 **	0.2	0.2	n/a
Total Phosphorus (mg/L)	0.13	0.075	0.02	0.02	0.04	0.04	<0.01
Sulphate (mg/L)	24.4	50.5	-	400	63	63	93.5
Manganese (mg/L)	0.08	0.14	1.9	0.1	0.52	0.3	0.132
Iron (mg/L)	0.71	0.70	N/A	0.3	0.31	0.7	0.11
Nickel (mg/L)	0.001	0.006	0.011	0.1	0.04	0.03	0.006
Zinc (mg/L)	0.005	0.006	0.008	5.0	0.08	0.02	0.007
Aluminium (mg/L)	0.18	0.135	0.055	0.2	0.07	0.07	<0.01
Copper (mg/L)	0.002	0.001	0.0014	1.0	0.0008	0.001	<0.001

The above results confirm that the levels at the edge of the mixing zone ANZECC trigger point was not exceeded for any analyte with the exception of Sulphate, although the level was well below the ANZECC default value. It should be noted that the SSTV for most analytes for the Wingecarribee River are actually lower than the ANZECC default triggers. These values are calculated using the 80th percentile of 24 monthly samples, that is, the level below which 80% of the readings fall. This is important to note because it means that water quality in the river itself, irrespective of the discharge from Berrima, will exceed the SSTV for about 20% of the time.

For the Wingecarribee River, these regular exceedances generally occur each time the river experiences very high flow and in some cases, very low flow during drought periods. During high flow events, the dilution of water from the Colliery discharge is over 1,000 times, in low flow periods in the river however, dilution may only be 1 or 2 times the discharge volume.

As previously discussed, there was a deterioration in water quality leaving the mine which has caused iron staining in the river just below the discharge point. The water monitoring results indicate that the elevated mineral concentration has not impacted the river at the Biloela site compared to historic discharge.

Based on water quality averages, the environmental values at Biloela have not altered. However, the higher mineral content discharged for a short period when the mine commenced free draining are still impacting the immediate mixing zone below the mine discharge.

3.6 Findings and Conclusions

Boral has implemented an underground water treatment system at Berrima Colliery to improve the water quality discharged into the Wingecarribee River. The treatment system is designed to increase the dissolved oxygen concentration and pH in order to precipitate Iron and Manganese. Precipitated metals are then removed by settlement prior to the water discharging off the site.

The system was trialled in late 2017 to early 2018 and was implemented by February 2018. The results to date have seen an improvement in most metals of concern, as well as a corresponding reduction in conductivity and sulphate. The treatment system has been proved successful and the discharge quality for most parameters are now the same or very close to historic discharge quality from the mine.

The results from ambient monitoring within the Wingecarribee River show a slight improvement however given the exceptionally dry conditions, the natural flow in the river has been very low. The discharge from the mine has been the dominant water source for the river for most of the past 12 months. Although large flow events also cause poor water quality, they also serve to remove the build-up of sediments as well as nutrients from surrounding farm land and the sewage treatment plant discharge.

Once the bulkheads are installed, water discharged from the mine will significantly reduce for a period of around 6 months. This will reduce the mineral load from the mine but also reduce the baseflow within the river which helps to support downstream river health.

4. Aquatic Ecology

4.1 Introduction

The ecological health of the Wingecarribee River is a component in the assessment methodology of the ANZECC 2000 guidelines. In order to understand the human impact on aquatic ecosystems, monitoring is progressively moving towards an approach using ecological measures compared to the traditional chemical indicators (Ravengai et al. 2005). Aquatic macroinvertebrates can be used as biological indicators because chemical characteristics including metal concentrations and acidity are driving their spatial distributions (Alvial et al. 2012). Macroinvertebrates are organisms visible to the naked eye that do not possess a backbone, including insects and larvae, worms and shrimp (Barbour et al. 1992).

Macroinvertebrates are positive indicators of stream health because they are frequent, widespread and sedentary in nature (Basset, Pinna & Renzi, 2017). Different species are sensitive to a variable range of pollutants, water quality and habitat conditions which helps pinpoint threats within the waterway (Barbour et al. 1992). They have relatively short lifespans, from a few months to years, which allows efficient detection of changes to water quality (Basset, Pinna & Renzi, 2017). Macroinvertebrates exist at the base of the food web, allowing a greater understanding of bio-accumulation and risks associated with higher trophic levels (Sullivan et al. 2014)

Three individual aquatic ecology studies have been conducted to examine the impact of Berrima Colliery Mine discharge on the Wingecarribee River over a 6 year period. The first study was carried out by Marine Pollution Research Pty Ltd from 2012 to 2014 (MPR), followed by a study by Wright, Paciuszkiewicz and Belmer in 2017 (Wright) and finally a third study by Niche Environment and Heritage in 2018 (Niche). These studies shared a common aim to evaluate how aquatic macroinvertebrate populations respond to mine discharge over a distance gradient along the Wingecarribee River, in order to assess the overall health of the ecosystem.

4.2 Methods

The studies sampled at variable locations, ranging from 3500m upstream of the discharge point, to 7000m downstream on the Wingecarribee River. Although these sample sites are not parallel across all studies, they can be compared on a gradient to observe changes along the mixing zone of the river. Figure 4.1 shows the locations of the sites sampled in the three studies.

Figure 4.1 Location of sample sites

Each study used comparable macroinvertebrate sampling techniques known as riffle sampling (MPR), kick sampling (Wright) and dip net sampling (Niche). Such techniques involve holding a 30cm wide net on the riverbed into the flow of the water and disturbing the upstream substrate with the feet. Wright and Niche timed each sample for 30 seconds and one minute respectively, whereas MPR disturbed the substrate for a length of 10 metres per sample. Samples were taken in riffle habitats, defined as an area of broken water with rapid current that has some cobble or boulder substratum (MPR, 2014). The number of replicates taken per study is variable, with seven, ten and five samples taken at each site by MPR, Wright and Niche respectively. Samples at all locations were always undertaken during the same or succeeding day. Samples are stored in containers and preserved with 70% ethanol whilst taken to the laboratory for taxonomic analysis to the family level.

All studies were undertaken during different years which enables the application of a metanalysis to observe the effects of the mine discharge over time. Table 4.1 below shows the dates of sampling for each study.

Table 4.1 Sampling dates of studies conducted by MPR, Wright and Niche

	Sample 1	**Sample 2**	**Sample 3**
MPR	November 2012	June 2013	May 2014
Wright	February 2017		
Niche	March 2018	June 2018	September 2018

The Wingecarribee River flow monitored approximately 3500m upstream of the discharge at Macarthur's crossing was recorded for sampling months as well as the volume of water released from the adit discharge point.

In order to measure the response of macroinvertebrates within the mine discharge mixing zone in the Wingecarribee River, comparisons have been drawn among three biotic indices. Measuring the macroinvertebrate population using taxonomic richness, abundance and EPT provides a strong indication of the ecological health of the river (Qu et al. 2010). Taxonomic richness refers to the number of different individuals observed, often performed at the family level. Abundance is a measure of the total number of macroinvertebrates included in the sample. The EPT index is used to calculate the abundance of pollution sensitive macroinvertebrates from the Orders Ephemeroptera, Plecoptera and Trichoptera (Wright & Ryan 2016).

4.3 Results

The river flow during the sampling months is recorded in Table 4.2 below. The majority of months had similar averages ranging from 7.4 ML/Day to 15.26 ML/Day. One outstanding month was June 2013 which had a monthly average of 418 ML/Day.

Table 4.2 Wingecarribee River flow during sample months

Study	**Period**	**Average (ML/Day)**	**Min**	**Max**
MPR	November 2012	15.26	3.18	143
	June 2013	418	24	2390
	May 2014	12.8	4.9	32
Wright	February 2017	13.3	1.85	80.4
Niche	March 2018	8.16	1.9	31.5
	June 2018	14.4	6.35	34.9
	September 2018	7.4	2.3	13.9

The average, minimum and maximum V Notch discharge volumes (mine discharge) for the sampling months is recorded below in Table 4.3. The average monthly volume ranged from 2.05 ML/Day to 4.25 ML/Day. The volume discharged in September 2018 was notably higher than the other months sampled.

Table 4.3 V Notch Discharge Flow

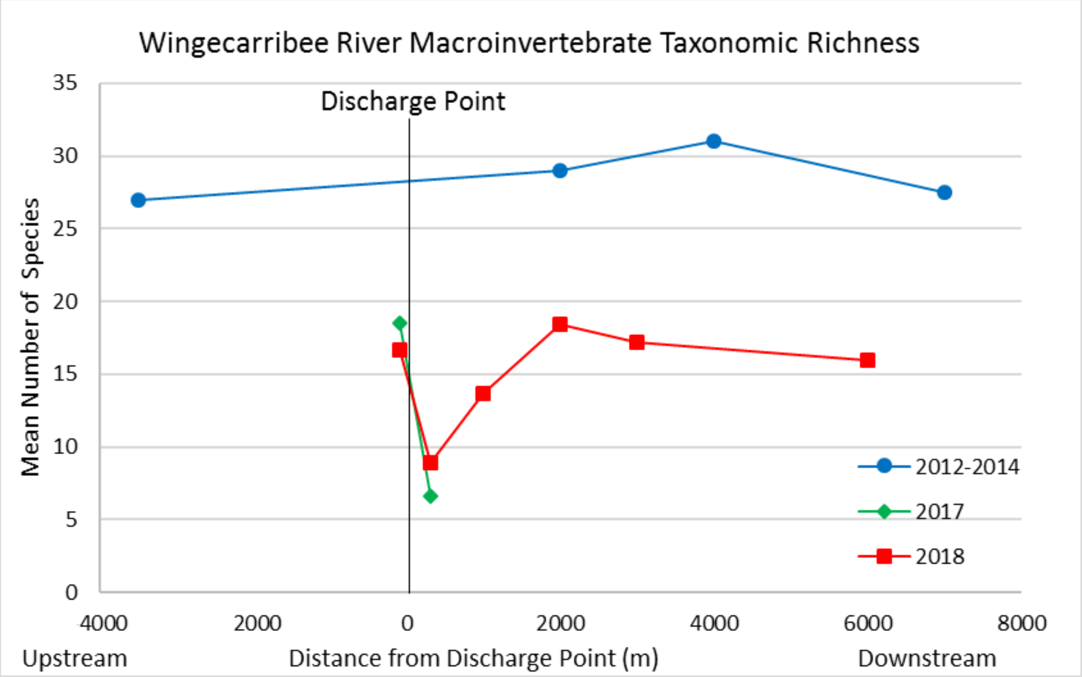
Study	Period	Average (ML/Day)	Min	Max
MPR	November 2012	2.78	0.77	3.51
	June 2013	2.78	0.32	4.05
	May 2014	3.28	1.45	4.63
Wright	February 2017	2.47	2.26	2.75
Niche	March 2018	2.29	1.97	3.38
	June 2018	2.05	1.49	2.33
	September 2018	4.25	1.64	5.14

4.3.1 Taxonomic Richness

Results show that the diversity of macroinvertebrate family assemblages has declined over time across the entire study area of the Wingecarribee River sampled. In 2012-2014, taxonomic richness ranged from 27 to 31 families with an average of 28.6, whereas the combined 2017 and 2018 data shows an average number of families was 14.5 with a minimum of 6.6 and a maximum of 18.5.

As seen in Graph 4.1, taxonomic richness dramatically declines by 36% and 53% in 2017 and 2018 samples respectively at 300m downstream of the discharge point. At the sampling point 1000m down, the data from 2018 shows that the macroinvertebrate diversity has increased to 13.7 families. By the 2018 sample at 2000m downstream, Graph 4.1 shows that a higher number of families exist at this location than samples 100m upstream of the discharge point, collected in the same year. A similar trend is present in 2012-2014, in which the lowest diversity is observed at the upstream site, and the three samples downstream of the adit discharge contain a greater range of macroinvertebrate inhabitants.

The most common type of macroinvertebrate varied between studies. In the study by MPR, insect taxa were the most common with 51 families, followed by 9 crustaceans and 8 molluscs. Meanwhile in 2017, the Aquatic worms *Oligochaeta* were sampled with the greatest frequency and the mayfly family *Baetidae* were second. Other families collected included a leech, freshwater mite, springtail, flatworm, hydroid and roundworm.

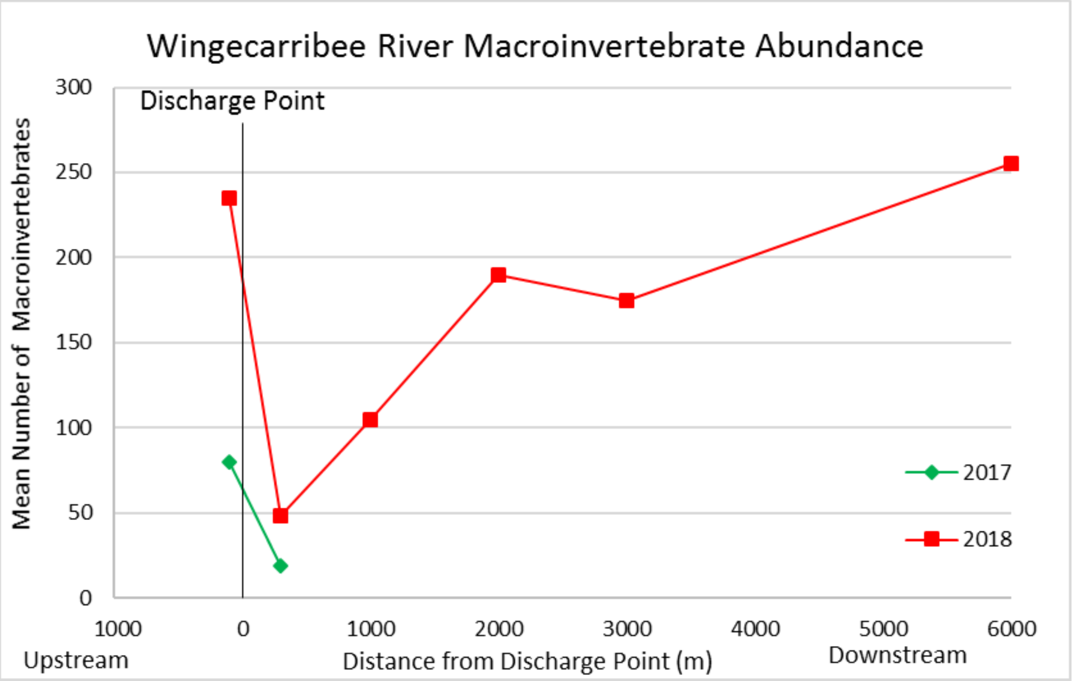


Graph 4.1 Macroinvertebrate Taxonomic Richness Upstream and Downstream of the Adit Discharge Point on the Wingecarribee River

4.3.2 Abundance

Aquatic macroinvertebrate abundance was sampled in 2017 and 2018 only. Graph 4.2 shows that the two samples collected in 2017 both had a lower abundance than the parallel sites in 2018. At 100m upstream of the discharge point, 80 macroinvertebrates were identified in 2017 compared to 235 in 2018. Similarly, the site 300m downstream showed 19 and 48 individuals sampled in 2017 and 2018 respectively.

As expected, both sampling years showed that the abundance was much lower immediately downstream of the discharge (300m) than at the site 100m upstream. In 2018, the abundance downstream increased with an increased distance from the discharge point. It is important to note that the furthest downstream point at 6000m was observed to have 20 more macroinvertebrates than the site 100m upstream of the discharge in 2018.



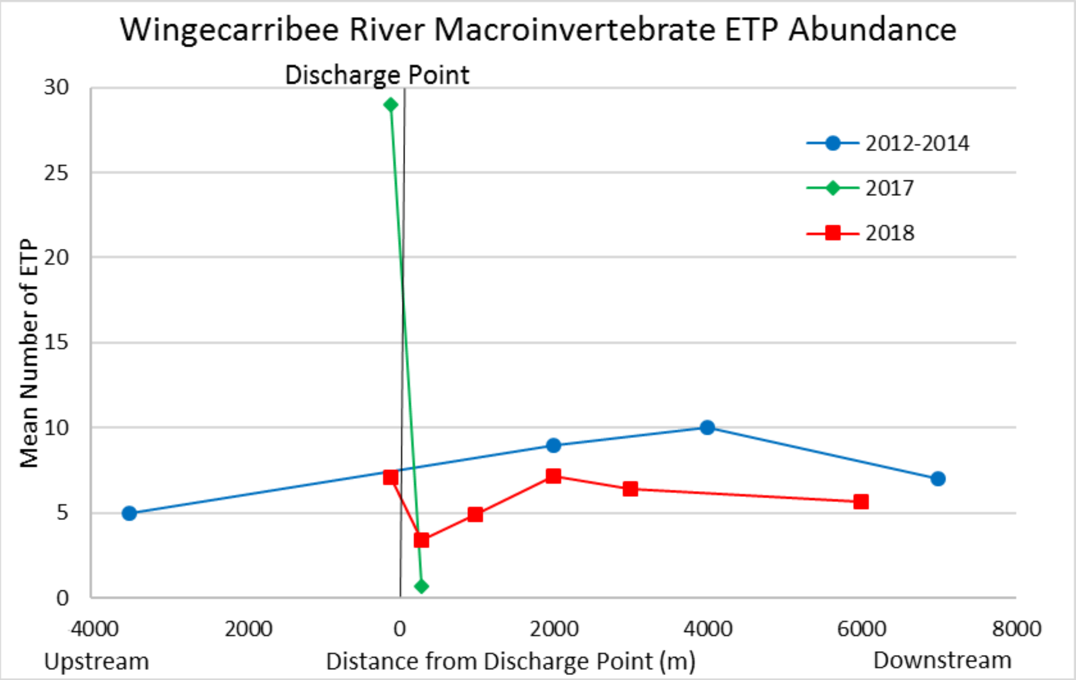
Graph 4.2 Macroinvertebrate Abundance Upstream and Downstream of the Adit Discharge Point on the Wingecarribee River

4.3.3 ETP Abundance

A substantially greater number of pollution sensitive EPT taxa was sampled at the 100m upstream site in 2017, with 29 macroinvertebrates from the orders *Ephemeroptera, Plecoptera* and *Trichoptera* identified. The study in 2017 showed the greatest range in abundance, from 29 at 100m upstream to 1 individual at 300m downstream. Data from 2012-2014 and 2018 were more comparable, with ranges of 5 to 10 and 3.4 to 7.1 respectively. Overall the 2012-2014 study observed a higher number of EPT taxa with a mean EPT abundance of 7.75 compared to a mean of 5.8 EPT in 2018.

Graph 4.3 shows that in 2012-2014, the number of EPT were always higher at downstream sites. Data from 2018 shows a decline at the 300m downstream site, followed by a steady increase to abundances similar to the site 100m upstream.

The 2012-2014 study found a total of 11 EPT taxa in the Wingecarribee River. An average of 8 of these families were found upstream, compared to an average of 7 located downstream. Similarly, 12 families from the EPT orders were identified in 2017; 8 in which were from the *Trichoptera* order, *Ephemeroptera* had 3 families present and the remaining family belonged to the order *Plecoptera.* All except for two of the *Trichoptera* families were present in 2018.



Graph 4.3 Macroinvertebrate ETP Abundance Upstream and Downstream of the Adit Discharge Point on the Wingecarribee River

4.4 Discussion

4.4.1 Variation Observed on a Temporal Scale

The results show that temporal variability is present between studies. In 2012-2014, MPR sampled a substantially greater number of macroinvertebrate families than sampled in 2017 or 2018. The abundance of macroinvertebrate both upstream and downstream of the discharge point was deemed greater in 2018 than 2017. The number of sensitive EPT individuals showed a dramatic decline below the discharge in Wrights' 2017 study, which was not reciprocated in 2012-2014 or 2018. There are several explanations to explain such differences, including the differences in sampling techniques and changes in river quality, which are further discussed below.

The method adopted by MPR in 2012-2014 involves sampling 10 metres for each replicate, compared to a stationary timed sample used in 2017 and 2018. This technique may result in a greater number of taxa being collected because an increased number of microhabitats including multiple boulders and reeds are sampled. Abundance was lower in 2017 than 2018, which could be due to the extended length of sampling in 2017 that allows macroinvertebrates to escape the net.

An alternative explanation is that more families were detected by MPR in 2012-2014 due to higher habitat quality. These samples were taken when the mine was still in operation, discharging water with lower mineral content. Coal extraction operations stopped in October 2013, underground pumping stopped, and the workings flooded with groundwater. While the mine was flooding, the discharge to the river was minimal. The mine commenced to free drain at the end of March 2016 and the discharge volume gradually rose to an average of 2.4ML/day and 2.5 ML/day during 2017 and 2018 respectively. Wright undertook his study 39 months after the Berrima Colliery had ceased mining, when groundwater was free-draining and the water quality was at its all-time lowest. The timing of the 2017 sampling can explain why total and EPT abundance was low. Underground treatment of water commenced in 2018, improving metal concentrations and overall river health. The 2018 results show increased taxonomic richness and abundance which supports the success of the water treatment.

A valid argument suggested by Loayza-Muro et al. (2010) is that taxonomic richness and abundance will not always be consistently high in the same samples due to competition and community composition. As chemical concentrations are reduced, more sensitive species will become non-existent from a site, hence reducing species richness and abundance. In turn, the resistant taxa will boost their abundance to replace sensitive species because the competition for resources is now limited. Only when concentrations become higher than the resistance for all species will the richness and abundance both decline (Loayza-Muro et al. 2010).

4.4.2 Impacts of Metals and Acidity on River Macroinvertebrates

A range of discharge elements can have variable deleterious effects on aquatic macroinvertebrates to cause reductions in population size and diversity. Slightly lowered pH values, increased iron, zinc and nickel are characteristic of the Wingecarribee River in 2017 and 2018.

Concentrations of zinc have been known to reduce energy absorption in macroinvertebrates, which has been linked to a reduction in reproductive output (Ravengai et al. 2005). Iron precipitates in the water can cause the destabilisation of benthic substrate and destroy ecosystem habitat (Rodrigues & Bueno, 2016).

Low concentrations of metals including zinc and iron within waterways can cause water acidification (Ravengai et al. 2005). Acidity weakens shells and exoskeletons of macroinvertebrates (Stumpf, Darby and Gwilliam, 2009). Acidification can cause the addition of sulfuric acid and iron sulfate which consumes oxygen and limits respiratory efficiency and causes deformities in larvae (Rodrigues & Bueno, 2016).

Theses studies however usually refer to much greater pH lowering and metal concentrations than occur within the mixing zone. As shown in Chapter 3, the concentration of minerals below the mixing zone are now comparable to the levels in 2013 when the mine was operational. This mineral load has been a feature of the Wingecarribee River for nearly 90 years. By meeting the historic mineral load, it could be assumed that the impacts of the mineral composition of the discharge will be no greater. As the concentration of mineral elements generally meet ANZECC default trigger values for 95% ecosystem protection, it should not be a limiting factor to river health.

4.4.3 Impacts on Macroinvertebrate Downstream of the Discharge Point

The taxonomic richness, total and EPT abundance showed significant declines at 300m downstream of the discharge in 2017 and 2018. These results show that the groundwater released from the mine in recent years has impacted on the macroinvertebrates within the mixing zone. No decline in abundance or richness was observed at a distance of 2000 m below the discharge in 2012-2014. Both the 2012-2014 and 2018 study show that below 2000 m the macroinvertebrate diversity and abundance is the same or greater than the upstream levels.

These results confirm that the river is negatively impacted close to the site of the discharge, however the distance of impact is short and quickly improves downstream. MPR suggest that the mixing zone required to return to healthy levels is approximately 1500m. The data collected downstream in 2012-2014 and 2018 are higher than upstream sites on multiple occasions, indicating a complete recovery. These observations conclude that no long term impacts on macroinvertebrate populations are evident.

The return of the river to the equivalent of background ecological function and health meets an overriding objective of the ANZECC 2000 guideline. The poorer mixing zone ecology in 2018 compared to 2013 should improve over time assuming that the discharge quality remains similar to historic levels.

4.4.4 Differences in Species Diversity and Abundance

The 2012-2014 study concluded that insect larvae were the most common macroinvertebrates and in opposition, worms were the most frequent in 2017. Samples were taken in different seasons, which could result in the presence of different species. The concentration of different pollutants at different time periods can also be linked to differences in species composition.

Samples by MPR were taken in spring and winter, whereas Wright undertook his study in summer 2017. Many species-specific explanations can be assumed, such as reproductive seasons, periods of torpor as experienced in molluscs, and times of higher food availability. High diversity is known to occur in periods of low water flow and increased temperatures (Waterwatch, 2001), although results show that flow was relatively consistent during most of the sampling months. Hill, Sayer & Wood (2016) claim that the best season to evaluate macroinvertebrate diversity is during Autumn.

All organisms have altered tolerances to different contaminants. The higher toxicants present in 2017, could limit the abundance of some species, but increase the population sizes of others due to reduced competition (Loayza-Muro et al. 2010). A representative species of this is *Chironomidae*, the non-biting midge who have been observed to become dominant in locations of high metal concentrations (Smolders et al. 2003). Chironomids are known to benefit from a dynamic environment because they can adapt to change and recolonise when other species diminish (Ravengai et al. 2005; Qu et al. 2010).

In correspondence with Wrights' study, Qu et al. (2010) claims that *oligochaetes* known as worms are a contamination tolerant taxon that are often inhabitants of degraded waterways. *Oligochaetes* were also claimed to be metal tolerant by Loayza-Muro et al. (2010), which was also the case in 2017 when the Wingecarribee River had its highest metal contamination levels recorded.

Previous studies have also agreed with findings by MPR in 2012-2014, in which *Diptera* was the most common species in health aquatic ecosystems. Qu et al. (2010) found that *Diptera* are highly sensitive to Cadmium and Lead. Some groups of *Diptera* were also deemed sensitive to metals and acidity in another study (Loayza-Muro et al. 2010).

4.5 Conclusion

The combined results of the studies presented above support the contention that the mine discharge into the Wingecarribee River does influence macroinvertebrate diversity and abundance below the mixing zone. In terms of aquatic ecology, the mixing zone is approximately 2000 m downstream of the discharge point. The studies have also shown that a complete recovery is observed further downstream. Macroinvertebrate populations were observed at their lowest existence in 2017, however water treatment commenced in 2018 which have seen an increase in population health that will continue into the future.

Under the Performance Monitoring Program specified on the Environment Protection Licence, further aquatic ecology studies will be undertaken in 2019. This additional data will be reported in February 2020.

5. Ecotoxicology Assessment

5.1 Introduction

Ecotoxicology refers to the study of the effects of chemical interactions on organisms (Assessment of Ecotoxicity, 2014). Such studies are common in freshwater ecosystems due to the discharge of waste into natural waterways (Assessment of Ecotoxicity, 2014). The goal of ecotoxicology is to understand and predict the impacts of pollution to avoid any damaging costs to ecosystems, and to restore previously polluted environments (Altenburger, 2011). Common pollutants to waterways analysed in ecotoxicology studies include Heavy Metals, PCBs, Pesticides, Mould and Chlorine (Chapman, 2002).

Ecotoxicology studies will examine the impact on a species as well as the compounding risks to ecological entities (Segner, 2011). This knowledge is important because exposure to toxic chemicals can have detrimental effects on species at an individual and community level. Chemicals which are absorbed by organisms in high doses can cause death directly, or indirectly cause population decline via changes in behaviour, reproduction, development and mutation (Relyea and Hoverman, 2006). The cascading effects are clear, with lower trophic level declines reducing food sources and biomagnification of chemicals influencing predatory species.

Aquatic ecosystems have a high biodiversity, so indicator organisms are used to assess the toxicity of various hazards present in waterways (Assessment of Ecotoxicity, 2014). From this perspective, scientists can examine the transfer pathways of hazards from indicator organisms and how they are integrated into the ecosystem (Segner, 2011). Ecotoxicity is often measured via length of exposure, and endpoints used are often related to survival, growth, development, and reproduction (Peake and Tremblay, 2016). Studies will then develop triggers to predict when waterways can be classified as an ‘environment that may be affected’ (EMBA) (Relyea and Hoverman, 2006).

Ecotoxicological studies were conducted in the Wingecarribee River in 2012 and 2017/2018 to examine the influence of the Berrima Colliery discharge on ecosystem health. This study aims to:

- ☐ Identify potential chemical hazards at toxic levels within the Wingecarribee River during two time periods, and
- ☐ Compare changes in river toxicity from 2012 to 2017/2018.

5.2 Methods

The ecotoxicological studies were undertaken during the same time periods as the Aquatic Ecology study, that is, 2012 and 2017/2018 but were also undertaken at different locations. Comparisons will be made on a gradient along the mixing zone of the river.

Four samples from each monitoring site were sent to laboratories at Ecotox Services Australia (ESA) in 2012 and the OEH Environmental Forensics (EF) Team in 2017/2018 for assessment of toxicological quality. Sampling, sample handling and delivery followed laboratory recommended protocols.

Differing indicator species were used, however both studies used *C. dubia* and *P. australiensis*, and thus these are key species used for comparison in this study. Species have variable tolerance thresholds and therefore it must be noted that acute toxicity testing durations vary between species. The information for each species and sampling period is provided below in Table 5.1.

Table 5.1 Species Sampled Within Each Time Period and Duration of Acute Toxicity Testing

Sampling Periods	**Aug 2012**	**Nov 2012**	**Dec 2012**	**Dec 2017**	**Apr 2018**
Sampling Locations:	400m Up, Discharge, 100m dn, 200m dn			100m Up, 100m dn, 2000m dn, 2500m dn	
Ceriodaphnia dubia	48h	-	-	7d	24h, 48h, 6d
Parayta australiensis	-	96h	-	24h, 48h, 72h	96h
Chironomous tepperia	-	-	48h	-	-
Melanotaenia duboulayi	-	-	-	24h, 48h	24h, 48h
Hydra vulgaris	-	-	-	24h, 48h, 72h, 96h	24h, 48h, 72h, 96h
Raphidocelis subcapitata	-	-	-	72h	72h

Samples collected from the Wingecarribee River were diluted at different factors using filtered and thiosulphate-treated Sydney mains water with the addition of 5% mineral water, conductivity adjusted to 500 ± 20 µS/cm with filtered seawater. Four different dilution factors were used among species and sampling periods (Table 5.2).

Table 5.2 Four Treatment Dilution Factors Used

Dilution Factor 1	**Dilution Factor 2**	**Dilution Factor 3**	**Dilution Factor 4**
Control	Control	Control	Control
1.6	4	1	100
3.1	7	6.25	-
6.3	12	12.5	-
12.5	20	50	-
25	35	100	-
50	60	-	-
100	100	-	-

In-house cultures of each indicator species were added to Wingecarribee River water samples diluted to different factors, to test the toxicity of the river water. In 2012, the three study species were each subjected to acute toxicity testing of immobilization, compared to a wider range of chronic and acute endpoints used in 2017/2018 listed below:

- 7-d cladoceran *C. dubia* lethality and reproduction impairment (chronic)
- 72-h alga *R. subcapitata* growth inhibition (chronic)

- ☐ 48-h larval rainbowfish *M. duboulayi* imbalance (acute)
- ☐ 96-h shrimp *P. australiensis* lethality (acute) and
- ☐ 96-h 'pink Hydra' *H. vulgaris* lethality (acute).

Water chemistry was also tested to make predictions regarding which chemicals are likely contributing to water toxicity.

5.3 Results

A summary of the toxicity test results between the 2012 and 2017/18 is provided in Tables 5.3 and 5.4. It should be noted that direct comparison of the test results is difficult given these were done by different laboratories using different test methodologies. Although the test locations are similar, it is not possible to accurately determine the precise locations so the distances below the discharge points are indicative only.

Table 5.3 Summary of 2012 Toxicity Test Results

Test species (end point)	Upstream 400m	Discharge Point	Downstream 100m	Downstream 200m
C. dubia (immobilisation)	Mildly toxic at greater than 25% concentration	Acutely Toxic	Not Acutely Toxic	Not Acutely Toxic
P. australiensis (immobilisation)	Not Acutely Toxic	Not Acutely Toxic	Not Acutely Toxic	Not Acutely Toxic
C. tepperia (immobilisation)	Not Acutely Toxic	Not Acutely Toxic	Not Acutely Toxic	Not Acutely Toxic
M. duboulayi (imbalance)	Not Acutely Toxic	Not Acutely Toxic	Not Acutely Toxic	Not Acutely Toxic
H. vulgaris	Not Acutely Toxic	Not Acutely Toxic	Not Acutely Toxic	Not Acutely Toxic
R. subcapitata (growth inhibition)	Significantly lower cell yield	Significantly lower cell yield	Significantly lower cell yield	No difference

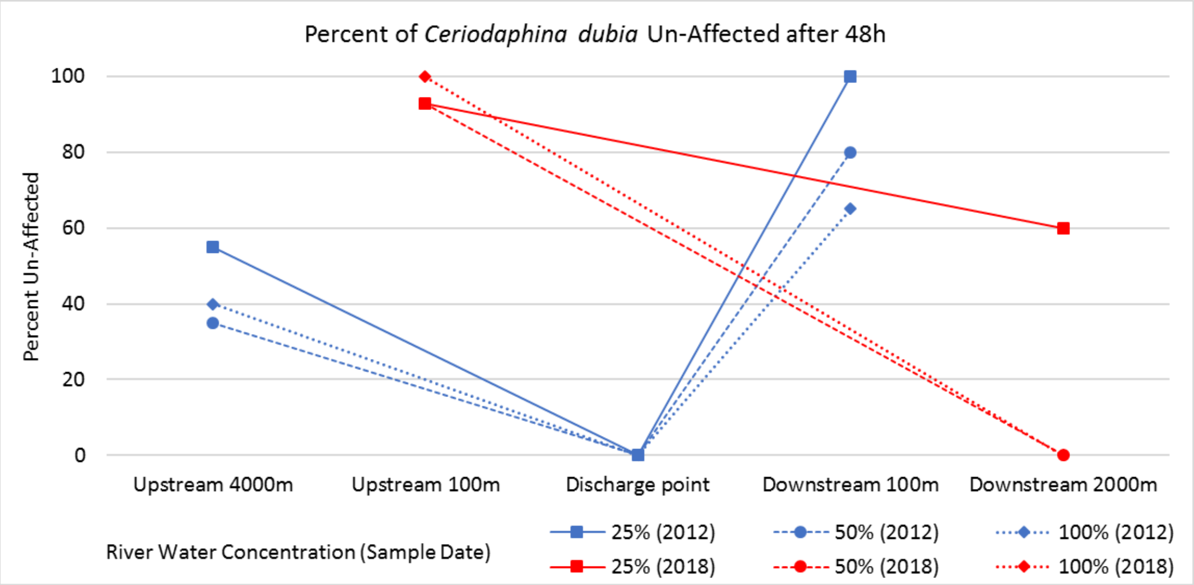
Table 5.4 Summary of 2017/2018 Toxicity Test Results

Test species (end point)	Upstream 100m	Downstream 100m	Downstream 2000m	Downstream 2500m
C. dubia (immobilisation)	-	Significantly lower survival and reproduction	No difference	No difference
M. duboulayi (imbalance)	Not Acutely Toxic	Not Acutely Toxic	Not Acutely Toxic	Not Acutely Toxic
P. australiensis (lethality)	Not Acutely Toxic	Not Acutely Toxic	Not Acutely Toxic	Not Acutely Toxic
H. vulgaris (lethality)	Not Acutely Toxic	Not Acutely Toxic	Not Acutely Toxic	Not Acutely Toxic
R. subcapitata (growth inhibition)	-	Significantly lower cell yield	No difference	No difference

The Graphs 5.1 to 5.6 below show a comparison of the main results observed for each indicator species at sites along the mixing zone, taken 5-6 years apart. Low dilution factors are not included in graphs because they are not tested on all occasions. Samples with a single point (Graph 5.4) indicate trials in which only 100% river water was compared against a control at only one site.

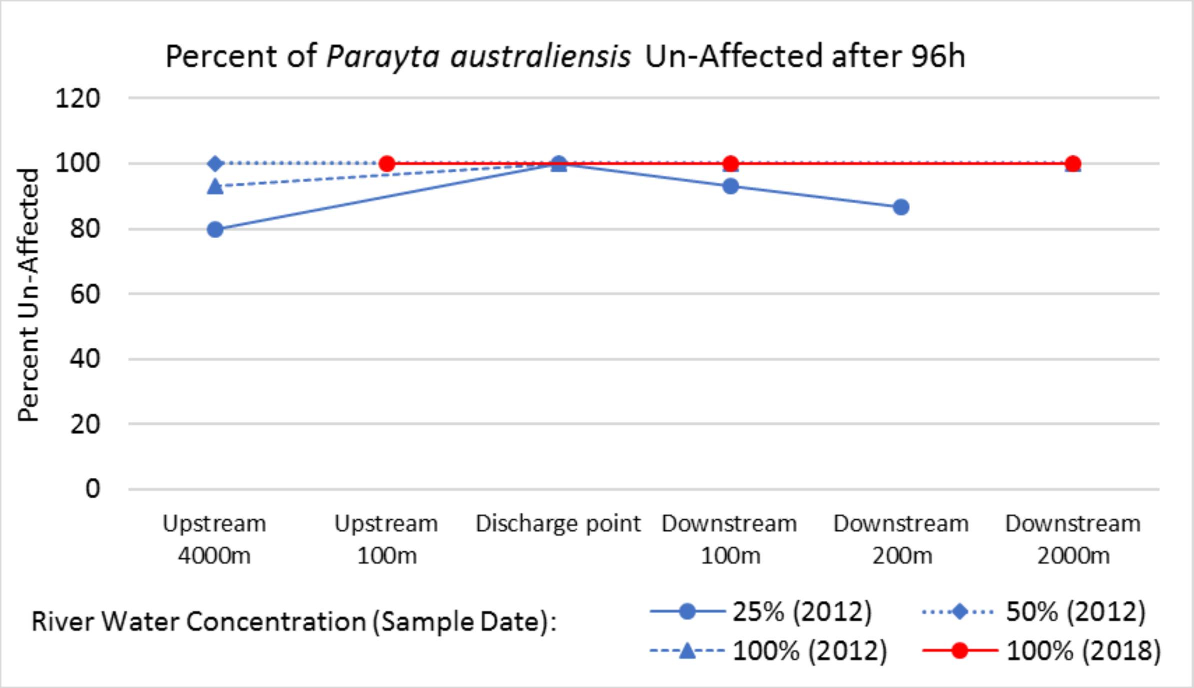
As expected, 100% concentrated river water caused a greater percentage of the water flea *C. dubia* to show effects from river water toxicants. In 2012, samples collected from the discharge point affected 100% of individuals. Graph 5.1 shows that in 2012 water at 4000m upstream had higher effects at 35 to 55% unaffected than 100m downstream of the discharge point at 65 to 100% unaffected.

Data from 2018 shows that water from 2000m downstream has significant effects with 0 to 60% survival and reproduction when compared to survival of 65 to 100% 100m upstream (Graph 5.1). An 11 times dilution was required to remove the toxicity to levels not causing an effect on *C. dubia* from 2000m downstream (see Table 5.5).



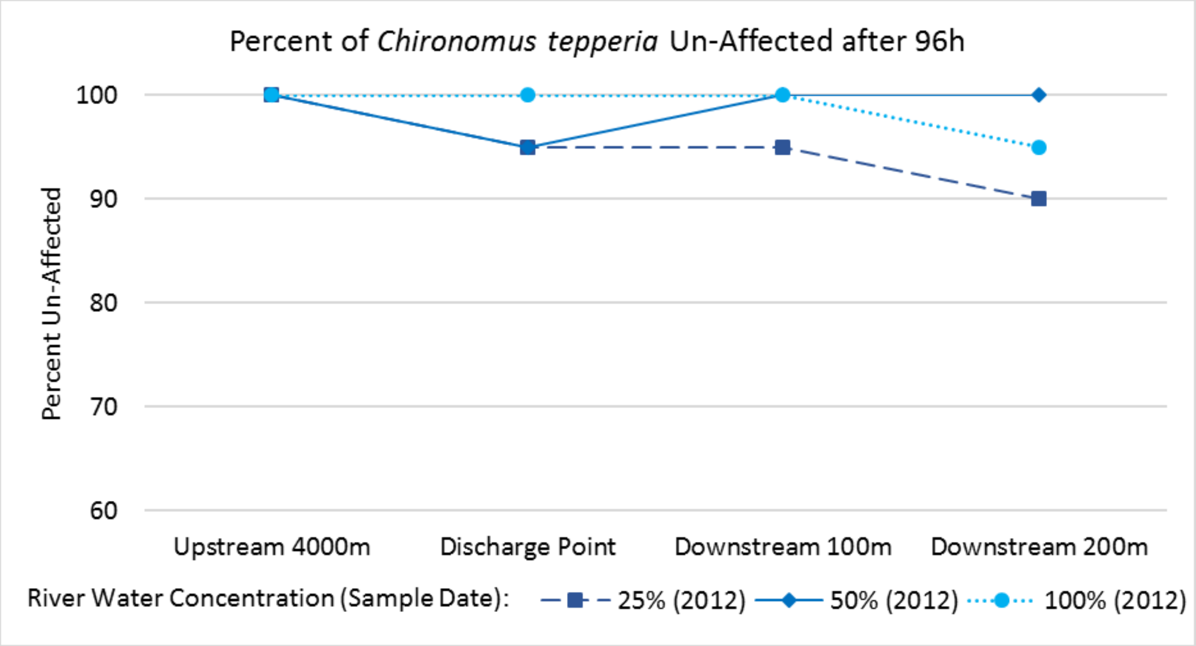
Graph 5.1 Percent *Ceriodaphnia dubia* unaffected after 48h

Graph 5.2 shows that only minor effects were experienced by the glass shrimp *P. australiensis* down to 80% unaffected at lower dilution levels of 25%. These effects in 2012 were not significant. Similarly in 2018, no acute toxic effects were observed in 100% concentrated waters at all sample locations. Samples taken in 2012 and 2018 were not significantly different at sites upstream or downstream of the discharge point.



Graph 5.2 Percent *Parayta australiensis* unaffected after 96h

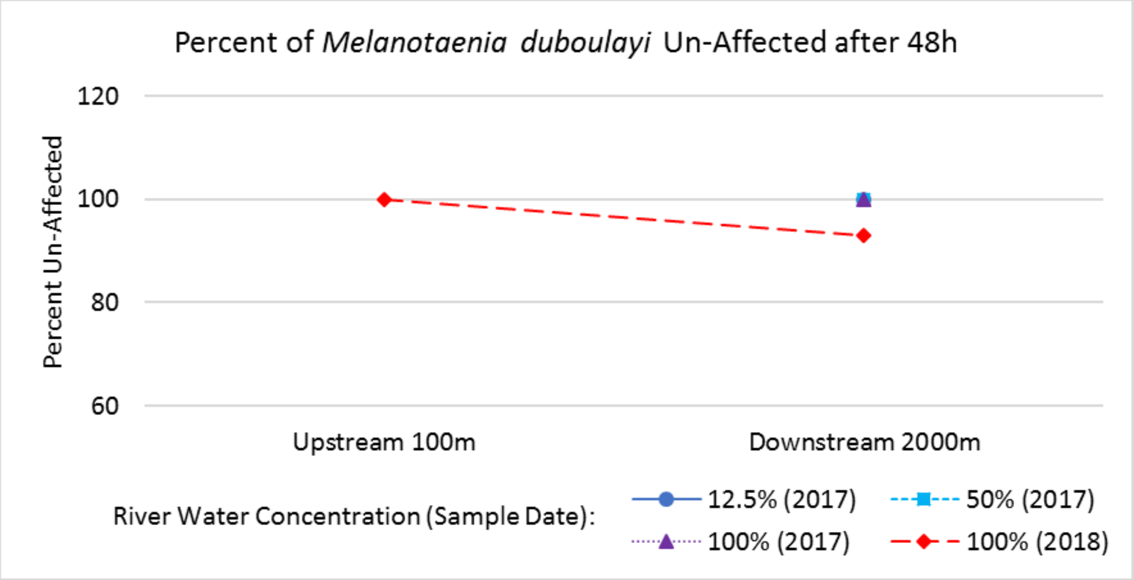
The non-biting midge (chironomid) larvae was used as an indicator in 2012 only. Percentages of unaffected individual remained above or at 90% at all dilutions and sampling sites. Results in Graph 5.3 indicate that there was no acute toxicity apparent for this species following 96 hours of exposure.



Graph 5.3 Percent *Chironomous tepperia* unaffected after 96h

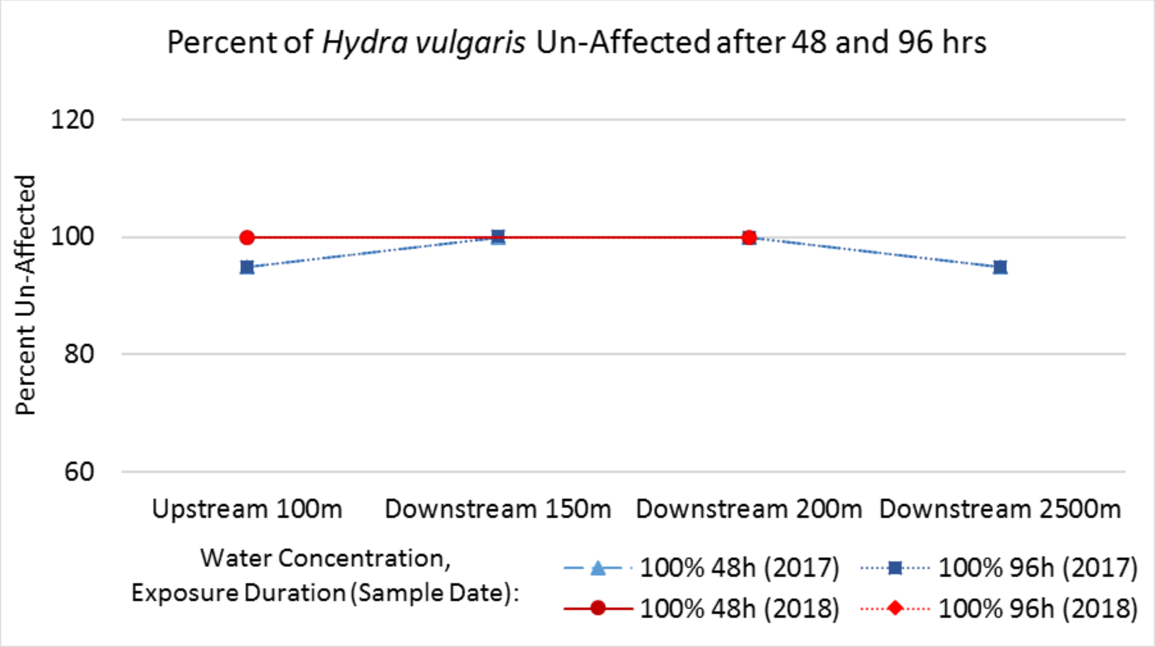
Graph 5.4 shows the 2017 and 2018 results of the larval rainbowfish *M. duboulayi* 100m upstream and 2000m downstream of the discharge point at three dilution factors. In 2017, all individuals were unaffected at 2000m downstream under three dilution factors; 12.5%, 50% and

100%. The percent of unaffected *M. duboulayi* dropped to 95% in 2018 at 2000m downstream, although this result was not deemed to have an acute toxic effect on the species.



Graph 5.4 Percent *Melanotaenia duboulayi* unaffected after 48h

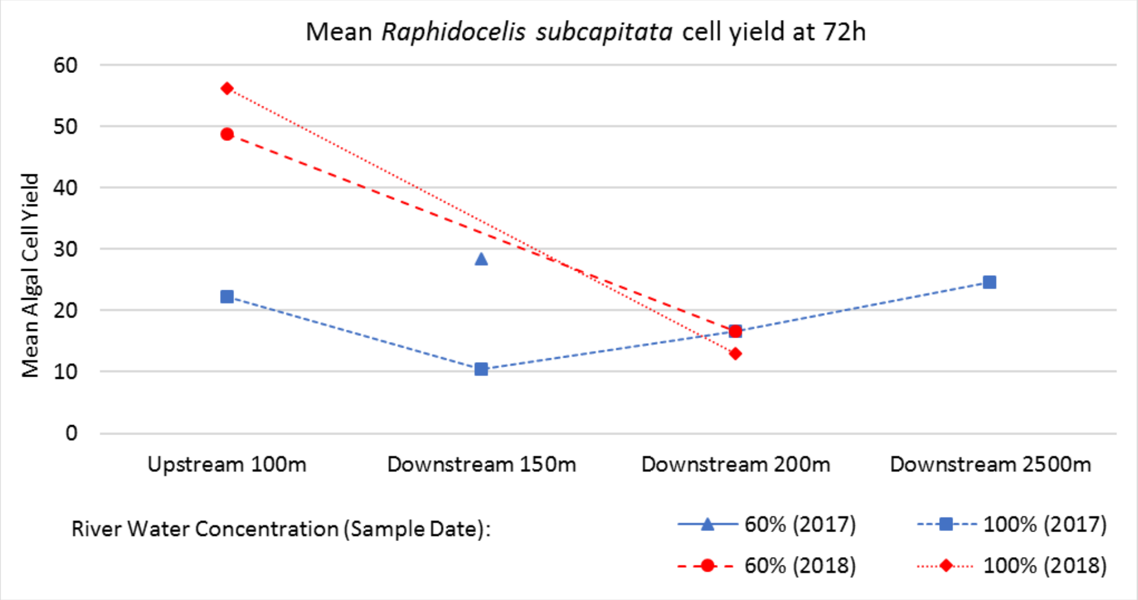
Pink hydra *H. vulgaris* was unaffected by river water at 100% concentration at all sites following 48 and 96 hours of exposure as seen in Graph 5.5. In 2017 5% of polyps were affected 100m upstream and 2500m downstream following 48 and 96 hours, however this was not a significant result.



Graph 5.5 Percent *Hydra vulgaris* unaffected after 48h and 92 hrs

Growth of the green microalga *R. subcapitata* was measured in 2017 and 2018 at 60% and 100% river water following 72 hours, as shown in Graph 5.6. Growth 100m upstream was 34% lower in 2017 than 2018 in undiluted river samples. Results from 2017 show that growth 100m upstream and 2500m downstream were at similar levels, with mean algal cell yields of 22.3% and 24.7% respectively. A 60% dilution of river water 150m downstream of the discharge resulted in an 18% increase in cell yield during 2017. Samples from 2018 saw 43% and 32% inhibition in algal growth in 100% and 60% river water samples respectively.

When compared to the control diluent (Table 5.5), sites 100m upstream, 200m and 2500m downstream exhibited an increase in algal growth. Undiluted water from 150m downstream were the only samples showing a decrease in algal cell yield relative to the control diluent.



Graph 5.6 Mean *Raphidocelis subcapitata* cell yield at 72h

5.4 Discussion

The results show that water discharged from underground effected the survival and reproduction of 100% of the water flea *C. dubia*, displaying both chronic and acute toxicity (See Table 5.5 for chronic results). No signs of toxicity were observed for the other indicator species tested at the discharge point; the glass shrimp *P. australiensis* or the midge larvae *C. tepperia*.

Hazardous influences of the Wingecarribee River water were not observed as close as 100m downstream in *C. dubia.* Water downstream of the discharge often displays similar ecotoxicity results as water upstream, suggesting that influences from the discharge do not extend into the mixing zone. In some cases, as for *C. dubia* and *R. subcapitata* (in 2012) the toxicity effects are higher at upstream sites, highlighting that external factors are also contributing to river health. It is also interesting to note that a higher percentage of the midge *C. tepperia* were impacted at sites furthest from the discharge point.

A chemical analysis prepared by OEH Environmental Forensics claimed that the chemicals likely contributing to the river toxicity included the metals Co, Ni, Zn and Mn, and the Ions Ca^{2+}, Mg^{2+}, and sulfate as SO_4. The presence of such ions have also attributed to slightly elevated conductivity and hardness within the downstream river sites.

Two of the indicator species had greater responses to river toxicity than the other four species. *Ceriodaphnia dubia* is a species of water flea known to be sensitive to low concentrations of metals (Hyne et al. 2005). This species has been a valuable asset to this study because it has shown disparity in response to pollutants at the different sites. The growth of the green microalga *R. subcapitata* was greatly reduced at 200m downstream, although growth was higher in water from all other sites than in the dilution control. A range of factors influence algae growth including light, carbon dioxide and nutrients, however a study by Baken et al. (2014) suggests that the presence of metals can limit nutrient availability.

In conclusion, the ecotoxicological studies undertaken in 2012 and 2017/2018 present conforming findings in regard to the toxicity of the Wingecarribee River along the mixing zone from the mine discharge. The water at the discharge point is deemed of acute and chronic toxicity for the sensitive cladoceran *C. dubia*. Other indicator species including the shrimp *P. australiensis,* chironomid *C. tepperia,* larval rainbowfish *M. duboulayi* and alga *R. subcapitata* showed no signs from discharge toxicity at any point within the immediate mixing zone.

Table 5.5

Ecotox Analysis for *Ceriodaphnia cf dubia*

Aug-12							
Location		U/S		Discharge		D/S	
Results	48hrs	% Un-Affected		% Un-Affected		% Un-Affected	
	Conc %	Av.	S.D	Av.	S.D	Av.	S.D
	1.6						
	3.1						
	6.3	100	0	100	0	100	0
	12.5	100	0	40	16.3	100	0
	25	55	10	0	0	100	0
	50	35	10	0	0	80	23.1
	100	40	23.1	0	0	65	25.2

Dec-17									
Location		U/S		D/S		D/S 2km		D/S 2.5km	
Results	7days	% Un-Affected		% Un-Affected		% Un-Affected		% Un-Affected	
	Conc %	Av.	S.D	Av.	S.D	Av.	S.D	Av.	S.D
	1.6			100	0				
	3.1			90	10				
	6.3			100	0				
	12.5			80	13				
	25	100	0	0	0	90	10	100	0
	50			0	0			90	10
	100	90	10	0	0	100	0		

Apr-18					
Location		U/S		D/S	
Results	24hrs	% Un-Affected		% Un-Affected	
	Conc %	Av.	S.D	Av.	S.D
	4	100	0	100	0
	7	100	0	100	0
	12	100	0	100	0
	20	100	0	80	0
	35	100	0	20	20
	60	93	7	7	7
	100	100	0	7	7

Apr-18					
Location		U/S		D/S	
Results	48hrs	% Un-Affected		% Un-Affected	
	Conc %	Av.	S.D	Av.	S.D
	4	100	0	93	7
	7	87	13	93	7
	12	100	0	93	7
	20	93	7	60	20
	35	100	0	53	29
	60	93	7	0	0
	100	100	0	0	0

Apr-18					
Location		U/S		D/S	
Results	6days	% Un-Affected		% Un-Affected	
	Conc %	Av.	S.D	Av.	S.D
	1.6	100	0	100	0
	3.1	100	0	100	0
	6.3	100	0	100	0
	12.5	100	0	100	0
	25	100	0	100	0
	50	100	0	100	0
	100	100	0	0	0

Ecotox Analysis for *Paratya australiensis*

Nov-12									
Location		U/S		Discharge		D/S		D/S	
Results	96hrs	% Un-Affected		% Un-Affected		% Un-Affected		% Un-Affected	
	Conc %	Av.	S.D	Av.	S.D	Av.	S.D	Av.	S.D
	6.3	100	0	100	0	100	0	100	0
	12.5	100	0	100	0	93.3	11.6	100	0
	25	80	34.6	100	0	93.3	11.6	86.7	23.1
	50	100	0	100	0	100	0	100	0
	100	93.3	11.6	100	0	100	0	100	0

Dec-17									
Location		U/S		D/S		D/S 2km		D/S 2.5km	
Results	24, 48, 72h	% Un-Affected		% Un-Affected		% Un-Affected		% Un-Affected	
	Conc %	Av.	S.D	Av.	S.D	Av.	S.D	Av.	S.D
	Control	100	0	100	0	100	0	100	0
	100	100	0	100	0	100	0	100	0

Apr-18									
Location		U/S		D/S		D/S 2km		D/S 2.5km	
Results	96hrs	% Un-Affected		% Un-Affected		% Un-Affected		% Un-Affected	
	Conc %	Av.	S.D	Av.	S.D	Av.	S.D	Av.	S.D
	Control	100	0	100	0	100	0	100	0
	100	100	0	100	0	100	0	100	0

Ecotox Analysis for *Chironomus tepperia*

Dec-12									
Location		U/S		Discharge		D/S		D/S	
Results	48hrs	% Un-Affected		% Un-Affected		% Un-Affected		% Un-Affected	
	Conc %	Av.	S.D	Av.	S.D	Av.	S.D	Av.	S.D
	6.3	95	10	100	0	100	0	95	10
	12.5	100	0	100	0	100	0	100	0
	25	100	0	95	10	95	10	90	11.6
	50	100	0	95	10	100	0	100	0
	100	100	0	100	0	100	0	95	10

Ecotox Analysis for *Melanotaenia duboulayi*

Dec-17									
Location		U/S		D/S		D/S 2km		D/S 2.5km	
Results	24hrs	% Un-Affected		% Un-Affected		% Un-Affected		% Un-Affected	
	Conc %	Av.	S.D	Av.	S.D	Av.	S.D	Av.	S.D
	1			100	0				
	6.25			100	0				
	12.5			100	0				
	50			100	0				
	100	100	0	100	0	100	0	100	0

Dec-17									
Location		U/S		D/S		D/S 2km		D/S 2.5km	
Results	48hrs	% Un-Affected		% Un-Affected		% Un-Affected		% Un-Affected	
	Conc %	Av.	S.D	Av.	S.D	Av.	S.D	Av.	S.D
	1			100	0				
	6.25			100	0				
	12.5			100	0				
	50			100	0				
	100	100	0	100	0	100	0	100	0

Apr-18					
Location		U/S		D/S	
Results	24hrs	% Un-Affected		% Un-Affected	
	Conc %	Av.	S.D	Av.	S.D
	Control	100	0	100	0
	100	100	0	100	0

Apr-18					
Location		U/S		D/S	
Results	48hrs	% Un-Affected		% Un-Affected	
	Conc %	Av.	S.D	Av.	S.D
	Control	100	0	100	0
	100	100	0	93	7

Ecotox Analysis for *Hydra vulgaris*

Dec-17									
Location		U/S		D/S		D/S 2km		D/S 2.5km	
Results	24, 48, 72h	% Un-Affected		% Un-Affected		% Un-Affected		% Un-Affected	
	Conc %	Av.	S.D	Av.	S.D	Av.	S.D	Av.	S.D
	Control	100	0	100	0	100	0	100	0
	100	95	5	100	0	100	0	95	5

Dec-17									
Location		U/S		D/S		D/S 2km		D/S 2.5km	
Results	96hrs	% Un-Affected		% Un-Affected		% Un-Affected		% Un-Affected	
	Conc %	Av.	S.D	Av.	S.D	Av.	S.D	Av.	S.D
	Control	100	0	100	0	100	0	100	0
	100	95	5	100	0	100	0	95	5

Apr-18					
Location		U/S		D/S	
Results	24, 48, 72h	% Un-Affected		% Un-Affected	
	Conc %	Av.	S.D	Av.	S.D
	Control	100	0	100	0
	100	100	0	100	0

Apr-18					
Location		U/S		D/S	
Results	96hrs	hrs		% Un-Affected	
	Conc %	Av.	S.D	Av.	S.D
	Control	100	0	100	0
	100	100	0	100	0

Ecotox Analysis for *Raphidocelis subcapitata*

Dec-17									
Location		U/S		D/S		D/S 2km		D/S 2.5km	
Results	72hrs	Algal Cell Yield		Algal Cell Yield		Algal Cell Yield		Algal Cell Yield	
	Conc %	Av.	S.D	Av.	S.D	Av.	S.D	Av.	S.D
	Control	13.4	0.5	13.4	0.5	13.4	0.5	13.4	0.5
	60			28.5	2.3				
	100	22.3	2.7	10.4	0.6	16.6	0.8	24.7	3.1

Apr-18					
Location		U/S		D/S	
Results	72hrs	Algal Cell Yield		Algal Cell Yield	
	Conc %	Av.	S.D	Av.	S.D
	1.6			24.1	3.9
	3.1			23.5	5.1
	6.3	51.3	3.9	28.3	5.3
	12.5	50.8	6.8	20.2	4.3
	25	36.8	6.3	23.8	3.9
	50	48.8	8.4	16.7	0.9
	100	56.3	4	13	1.1

6. Sediment Analysis - Fate Assessment

6.1 Introduction

The quality of sediment is important for the overall health of an aquatic ecosystem. Many aquatic macroinvertebrates and benthic organisms will obtain large amounts of their diet from the substrate, in which the levels of deposited contaminants consumed can have cascading effects on ecosystems (Chapman, 1992). This Chapter explains how the concentration of metals and other pollutants present in the sediment can influence macroinvertebrate health and ecotoxicity, as discussed in Chapters 4 and 5.

Sediment quality must be considered when drawing conclusions about the health of an aquatic ecosystem. Aquatic substrate is often an environmental sink for contaminants which undergo variable biological mechanisms including sorption, biodegradation and deposition which cause large amounts of particles present in the water column to end up in the benthic environment (Dickson *et al.* 1984). The river flow rate is directly related to the levels of pollutants present in the sediment (Chapman, 1992). High flow rates are associated with greater concentrations of total suspended solids, hence extending the length of the mixing zone and maintaining metals in solution.

Chemical reactions continuously influence the transition of metal distribution from the sediment to the water column, and back again (Chapman, 1989). Thus, it is important to analyse sediment quality in conjunction with the water quality to grasp the complete impact of the mine adit discharge on the health of the Wingecarribee River. Chapter 3 analyses the concentration of total and dissolved metals present in the river water, however these measurements only include the contaminants contained in the water column itself. In order to examine the concentration of deposited material, the sediment must be sampled to give a more complete picture of the river contaminants.

The analysis draws comparisons between sediment data collected 5 years apart, in 2013 and 2018, to view changes in the health of the river's benthic habitat on a temporal scale. This study aims to provide a comprehensive assessment of the contaminants present in all elements of the waterway, as well as tracking the movement of metals through deposition and river flow.

6.2 Methods

Several surface sediment subsamples are taken from multiple locations at each site to capture a clear representation of the site. Sediment is collected from the top 15cm of the substrate using a hand trowel and placed into 250ml glass jars supplied by Australian Laboratory Services. The samples were transported to the laboratory on the day of collection. Chain of Custody records are kept and include the time, date and location of sample, name of the person collecting the sample and records of the analysis to be performed. Water quality samples were taken a the same time and location on all occasions.

6.2.1 Ambient Sediment Quality Results

During 2013, sediment samples were taken on a monthly basis from the four ambient monitoring sites:

- ❒ Wingecarribee River upstream of the mine adit discharge at Old Hume Highway Crossing at Berrima, referred to below as Upstream
- ❒ Macarthur’s Crossing, also upstream of the mine but within the Hawkesbury Sandstone
- ❒ Wingecarribee River approximately 6km downstream of the mine adit discharge at Biloela Camp Site. Sampling commenced in August 2013.
- ❒ Wingecarribee River downstream of mine adit discharge at Black Bob’s confluence, referred to below as Downstream.

Samples were taken from the same sites in November 2018 to observe any changes in sediment quality over the past 5 years. Graphs 6.1 - 6.4 below show the results for Iron, Manganese, Nickel and Zinc monthly during 2013 and in November 2018.

6.2.2 Localised Sediment Quality Results

Sediment samples have also been taken in concurrence with the Performance Monitoring of the Wingecarribee River and an additional sample was taken in November 2018. These results give a more localised representation of the influence of the mine discharge on the river over shorter distances. Samples have been taken to test for the concentration of Iron, Manganese, Nickel, Zinc and Cobalt at the following sites:

- ❒ 100m upstream from the Adit Discharge (WR 100m Up)
- ❒ 300m downstream of the Adit Discharge (WR 300m Dn)
- ❒ 1km downstream of the Adit Discharge (WR 1km Dn)
- ❒ 2km downstream of the Adit Discharge (WR 2km Dn)
- ❒ 3km downstream of the Adit Discharge (WR 3km Dn)
- ❒ Biloela approximately 6km downstream (Biloela ~ 6km Dn)

6.2.3 River Load

In order to determine the fate of chemicals discharged from underground, water quality data was used to estimate mineral load, changes in concentration of minerals in sediments and to determine the effect of river flow. By multiplying the river flow taken from Macarthur’s Crossing by the water concentration of metals including Iron, Manganese, Nickel and Zinc, the load of metals expected to be travelling downstream as well as being deposited in the sediments was calculated. This allowed for a comparison of water quality changes to sediment variations over time and distance upstream and downstream of the discharge point.

Two sets of graphs are presented in the results below for each element tested. The first graphs show an intense analysis of three months showing the mineral load at every sampling site. The purpose of these figures is to show a detailed picture of changes over short distances along the river in order to pinpoint impacts from the discharge. The second set of graphs show monthly data, however data is not presented for all sites. This enables a wider view of the river with more flow and higher variability among data. Data is presented in two sets of figures to highlight the way different trends emerge by observing results in separate ways.

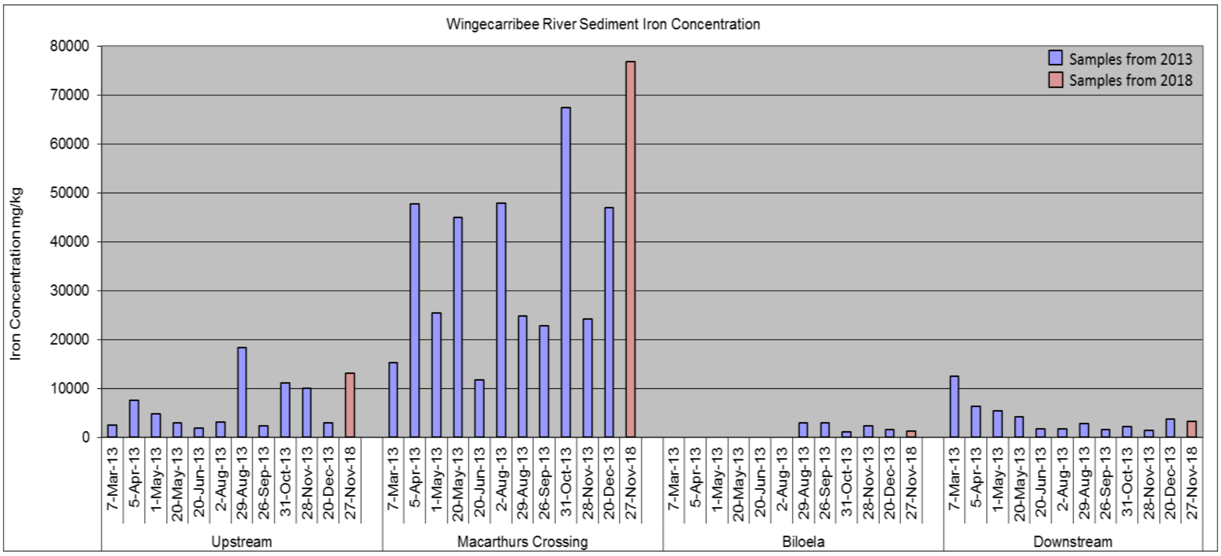
Tables 6.1 to 6.4 at the end of this Chapter show total quantity of minerals removed from the discharge (retained in the mine).

6.3 Results

6.3.1 Ambient Sediment Quality Results

Iron

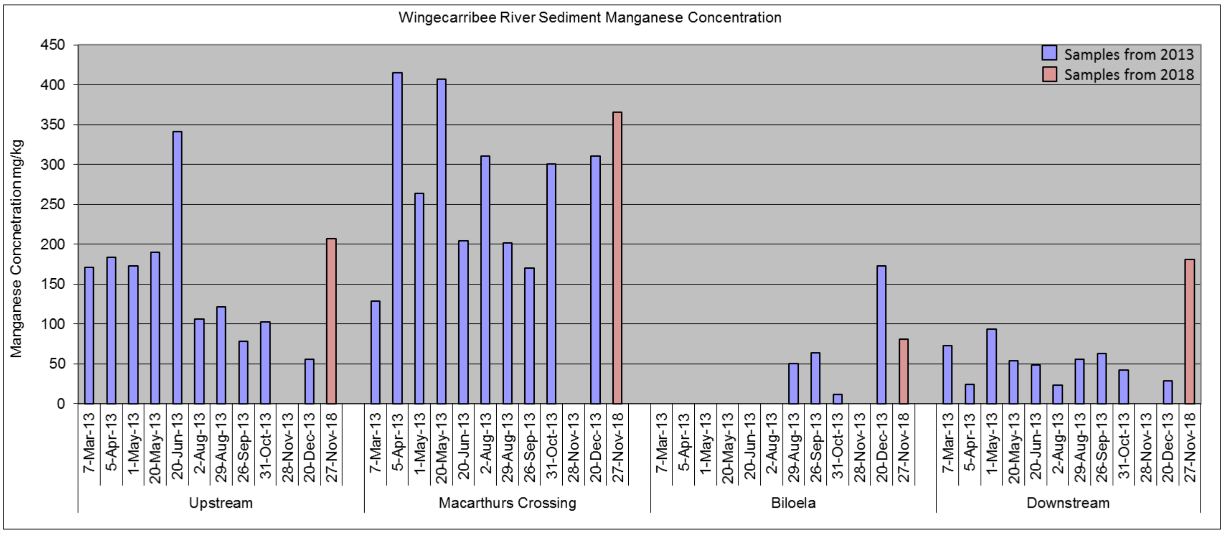
Graph 6.1 shows that the average Iron concentration at the Upstream site is 6750 mg/kg, while the average Iron was 38025mg/kg at MacArthur's Crossing. Average Iron levels at Biloela and Downstream were much lower, at 2085 mg/kg and 3966 mg/kg respectively. The notably high levels at Macarthur's Crossing are influenced by the Hawkesbury Sandstone geology located in this area. Iron was higher at Macarthur's Crossing in 2018 than 2013, however no difference between the two time periods was observed at the other three sampling sites.



Graph 6.1 Ambient Sediment Iron Concentration

Manganese

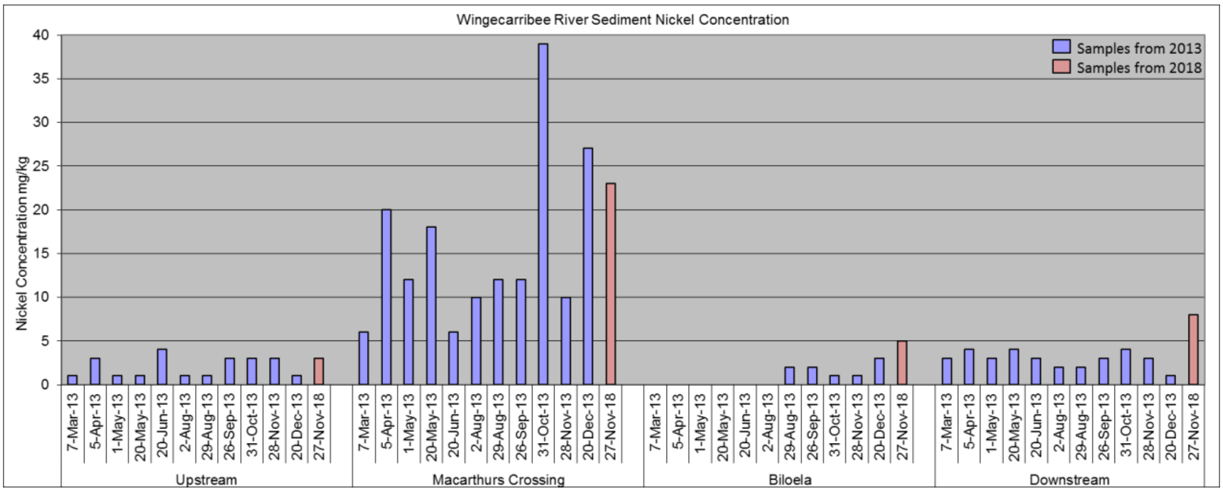
Similar trends in sediment concentrations are observed for Manganese in Graph 6.2. The presence of the Hawkesbury Sandstone at Macarthur's Crossing has caused an increased Manganese concentration of 280 mg/km compared to 76 mg/kg at Biloela. Manganese was higher in 2018 than the 2013 averages at the Upstream and Downstream sites.



Graph 6.2 Ambient Sediment Manganese Concentration

Nickel

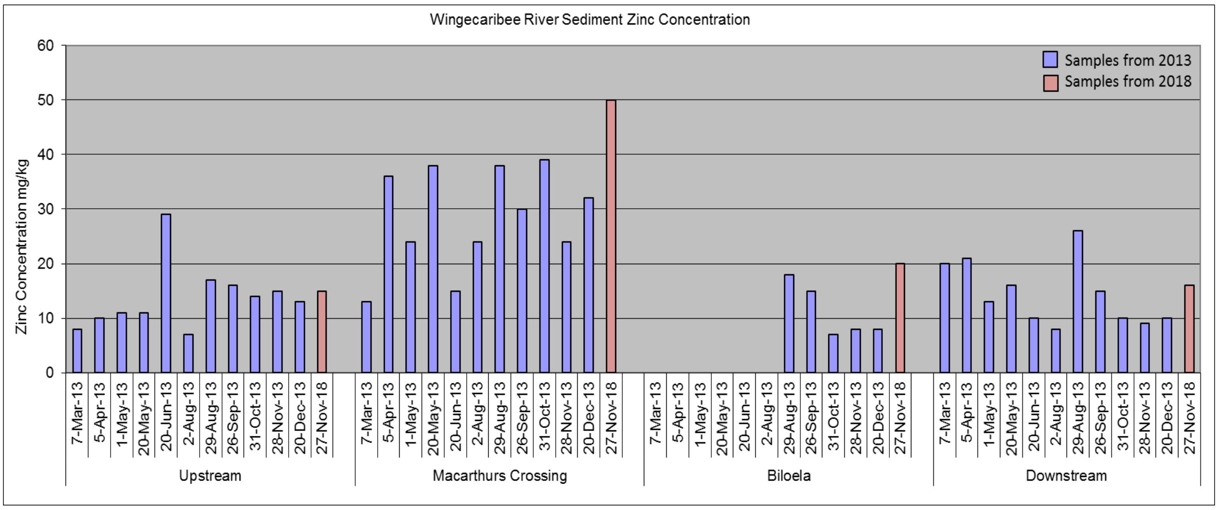
Again, the Macarthur's Crossing site had the highest sediment concentrations for Nickel due to its geology, with an average of 16.25 mg/kg. All other sites had average levels lower than 4 mg/kg. Levels of Nickel were slightly higher below the discharge point in 2018 compared to 2013.



Graph 6.3 Ambient Sediment Nickel Concentration

Zinc

The concentration of Zinc at Macarthur's Crossing was 30.25 mg/kg which is twice as high as the average concentration of Zinc at the other three sampling sites. From Graph 6.4 it can also be seen that the levels of Zinc at Macarthur's Crossing in 2018, reaching 50 mg/kg, were higher than in 2013. The concentration of Zinc below the discharge point at Biloela was slightly higher in 2018 compared with 2013, but were still well below the upstream site at Macarthur's Crossing.

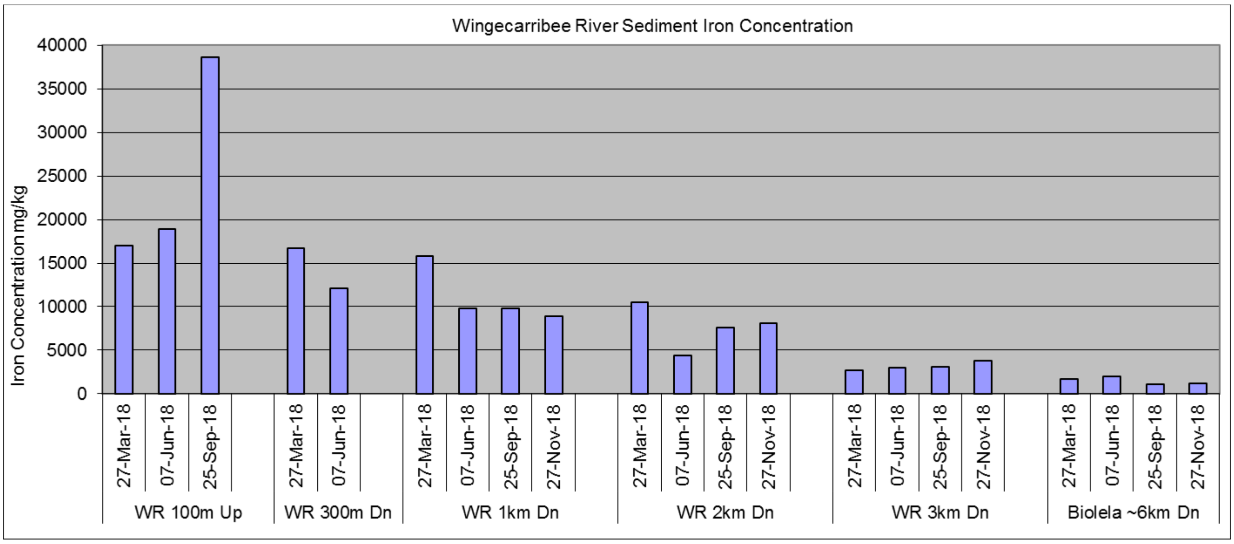


Graph 6.4 Ambient Sediment Zinc Concentration

6.3.2 Localised Sediment Quality Results

Iron

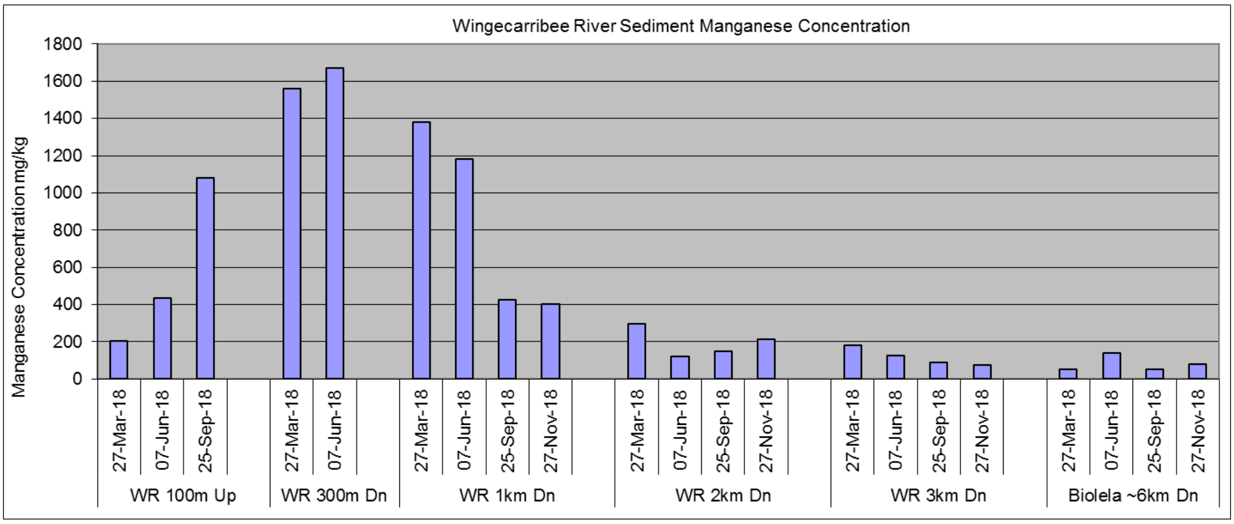
Graph 6.5 shows that the level of Iron contained in the river sediment gradually declines going downstream. The 2018 average Iron levels start at 24,833 mg/kg at WR 100m Up and drops to 1,515 mg/kg at Biloela. Since Iron is highest at the site above the adit discharge point, it shows that Iron is contained in the sediment from sources in addition to the discharge. There is visible evidence of Iron in the surface sediments just below the discharge point which gradually reduces with distance downstream. This gradual progression is shown below in Graph 6.5.



Graph 6.5 Localised Sediment Iron Concentration

Manganese

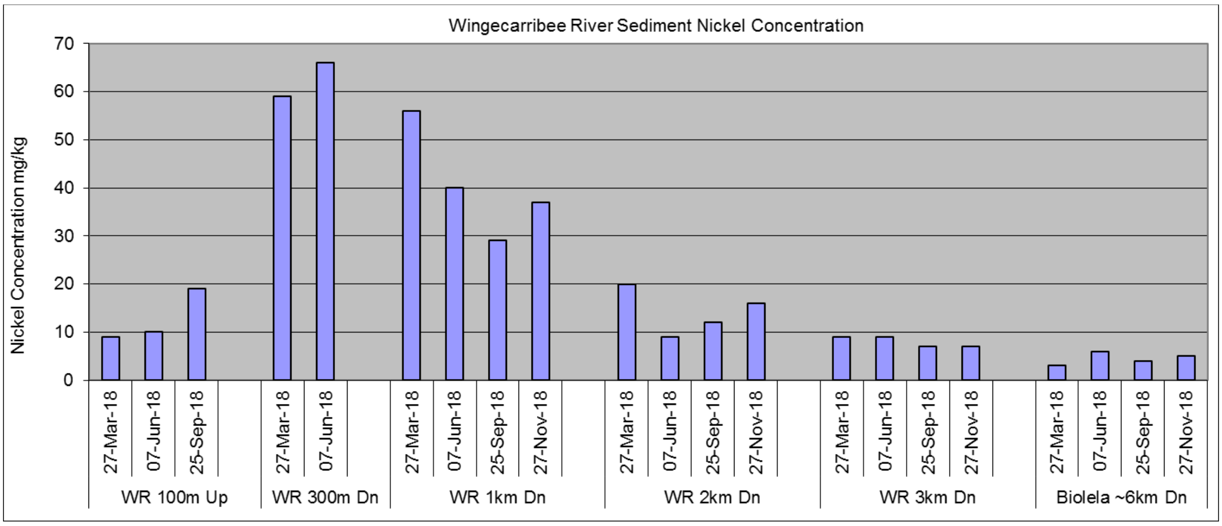
Manganese concentrations are highest at sampling sites WR ~300m Dn and WR 1km Dn, with average concentrations of 1,615 mg/kg and 847 mg/kg respectively. Levels decline progressively moving away from the discharge point, however levels are higher at WR 100m up than at WR 2km Dn, WR 3km Dn and Biloela. There is visible evidence of Manganese deposition within the mixing zone and the results of the sediment analysis correspond well with the water quality results from the same locations.



Graph 6.6 Localised Sediment Manganese Concentration

Nickel

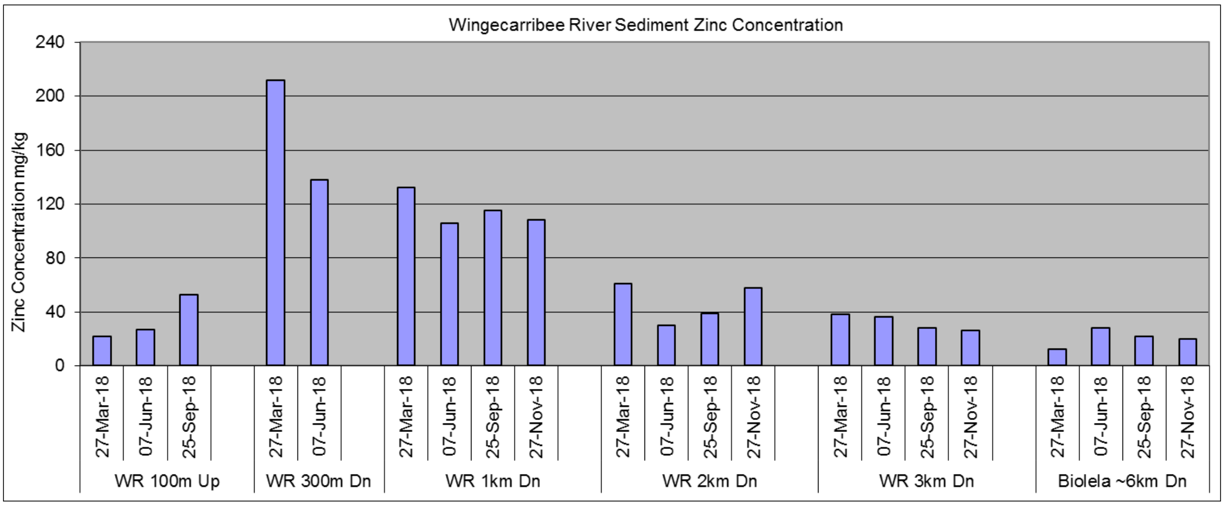
Graph 6.7 shows that sediment concentrations of Nickel follow a similar trend to Manganese levels. Nickel was highest in 2018 at WR ~300m Dn with an average of 62 mg/kg followed by WR 1km Dn with an average of 40 mg/kg. The four remaining sampling sites had averages below 15 mg/kg. These results also correspond to water quality results presented in Chapter 3.



Graph 6.7 Localised Sediment Nickel Concentration

Zinc

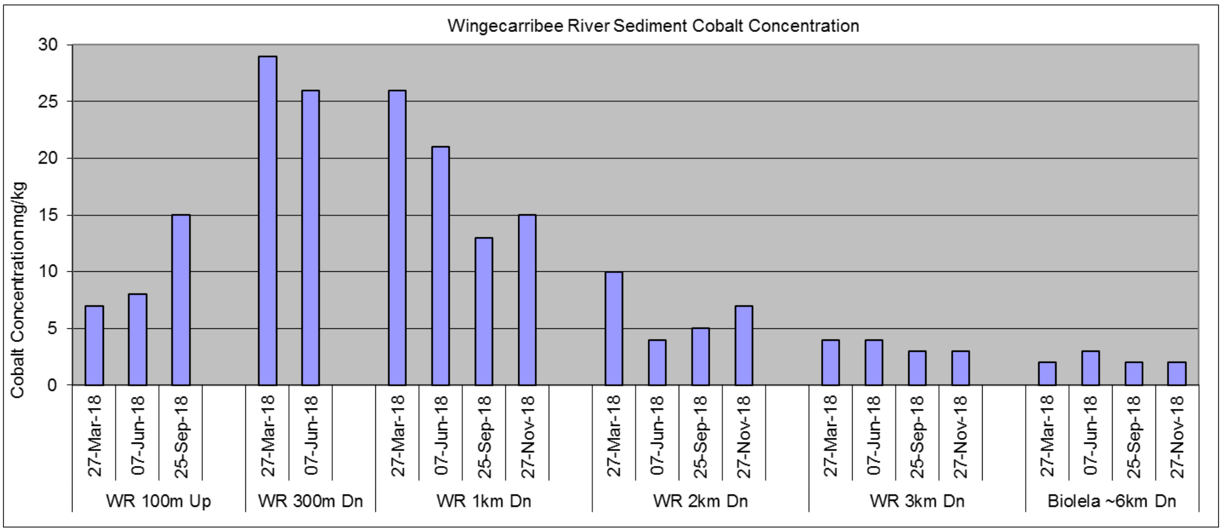
Graph 6.8 shows the Zinc concentrations at the Performance Monitoring sites during 2018. The average Zinc deposition at WR ~300m Dn was 175 mg/kg, while the average at WR 1km Dn was 115 mg/kg. The averages for the other four sites fell below 50 mg/kg. The variance in Zinc concentrations were higher than would have been expected from the corresponding water quality results which indicate that the Zinc has formed a stable sediment and is not bioavailable.



Graph 6.8 Localised Sediment Zinc Concentration

Cobalt

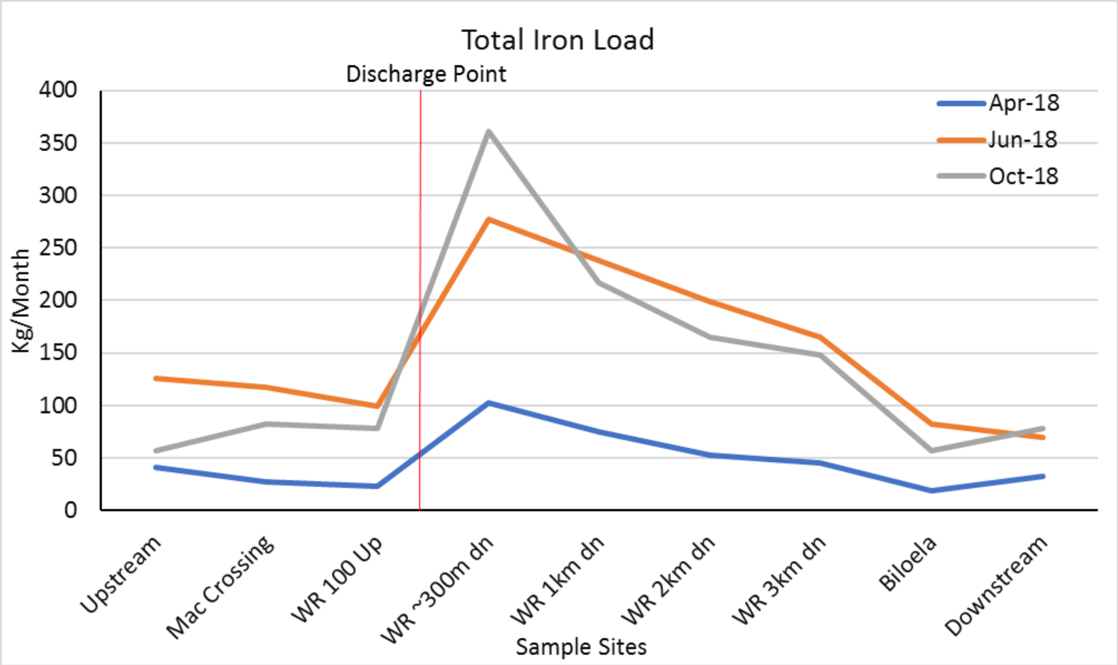
The same trends were observed for Cobalt as other metals, with averages at WR ~300m Dn and WR 1km Dn of 27 mg/kg and 19 mg/kg respectively, with Cobalt concentrations at these sites three and two-fold higher than the other sampling locations.



Graph 6.9 Localised Sediment Cobalt Concentration

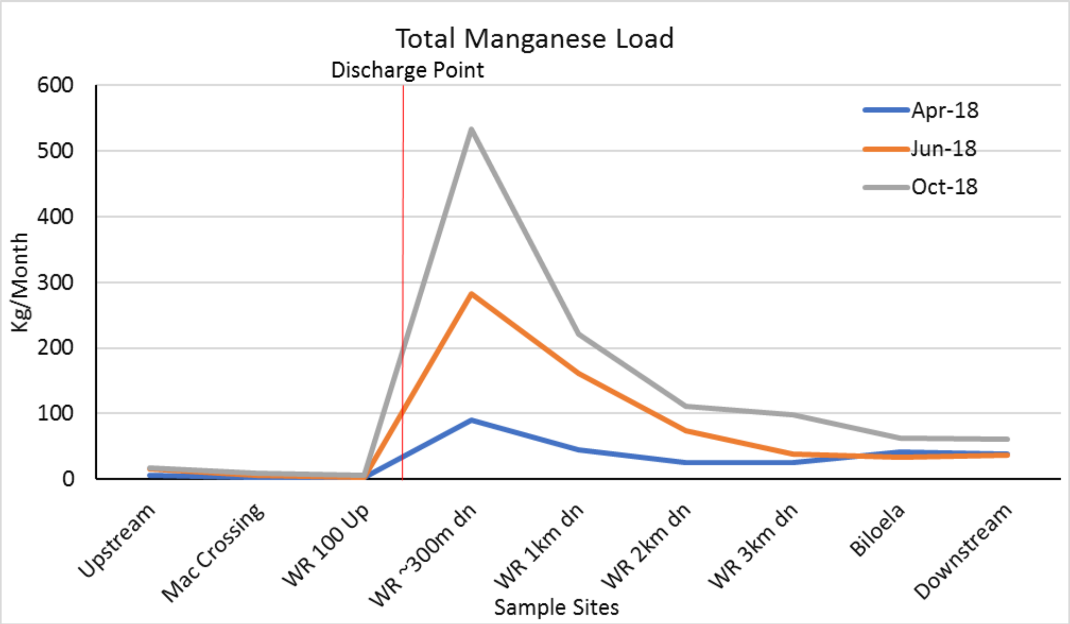
6.3.3 River Load

Graph 6.10 shows the total Iron load travelling down the Wingecarribee River over three months during 2018. All three sampling months saw an increased concentration of Iron at the site ~300m downstream of the discharge point, with a gradual decline to concentrations mirroring upstream levels at Biloela. Samples taken in April 2018 are consistently lower in Iron than those in June and October. October had the highest Iron load at WR~300m dn, although samples from June were highest at all other sites.



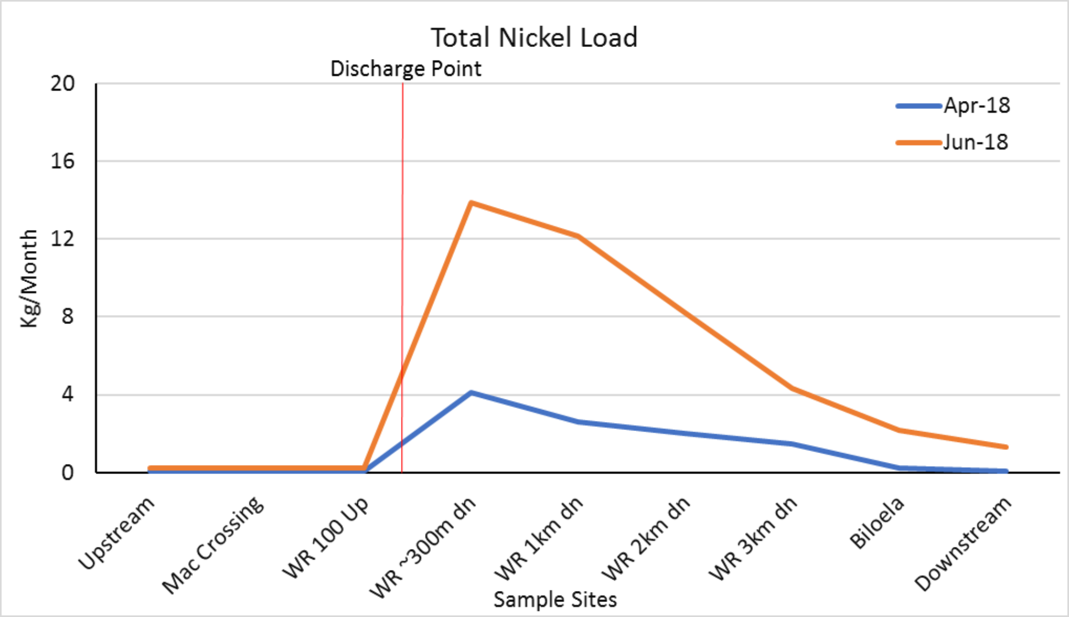
Graph 6.10 Total Iron Load in the Wingecarribee River at all Sites

The total Manganese load for 2018 is shown below in Graph 6.11. Concentrations of Manganese at WR ~300m dn progressively increased during the year by 190 kg/Month and 250 kg/Month. At sites further downstream, concentrations decline and the differences between months become smaller. Concentrations at the Downstream site remain 20 kg/Month to 30 kg/Month higher than at the Upstream site. This data indicates that the Manganese is precipitating and settling within the sediments immediately below the discharge point. This corresponds to the sediment analysis and water quality data. It also indicates that the Manganese is quite stable within the river sediment and would require a significant flushing event to transport the sediment load further downstream.

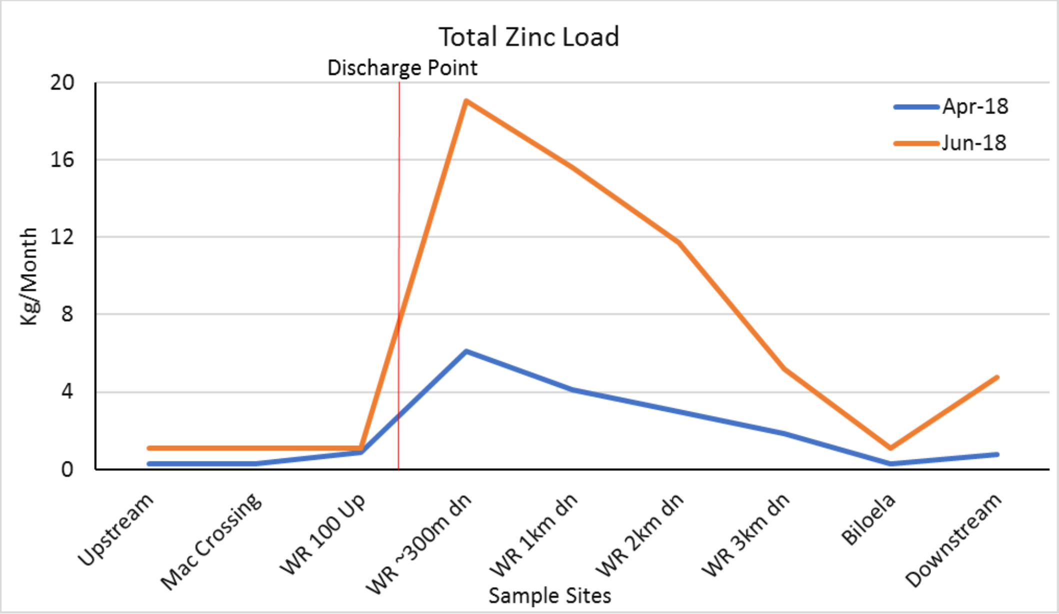


Graph 6.11 Total Manganese Load in the Wingecarribee River at all Sites

Similar trends occur for total river loads of Nickel and Zinc in April and June, shown in Graphs 6.12 and 6.13 respectively. Loads sit at or below 1.0 kg/Month at the three upstream sites, before rising dramatically at ~300m downstream. Nickel concentrations reached 4 kg/Month and 14Kg/Month in April and June, whereas Zinc loads were 6 kg/Month and 19 kg/Month in April and June respectively. Both Nickel and Zinc load concentrations had declined to levels similar to upstream sites at Biloela. An unexpected rise in Zinc was observed in June at the Downstream site which may be a result of high river flow.



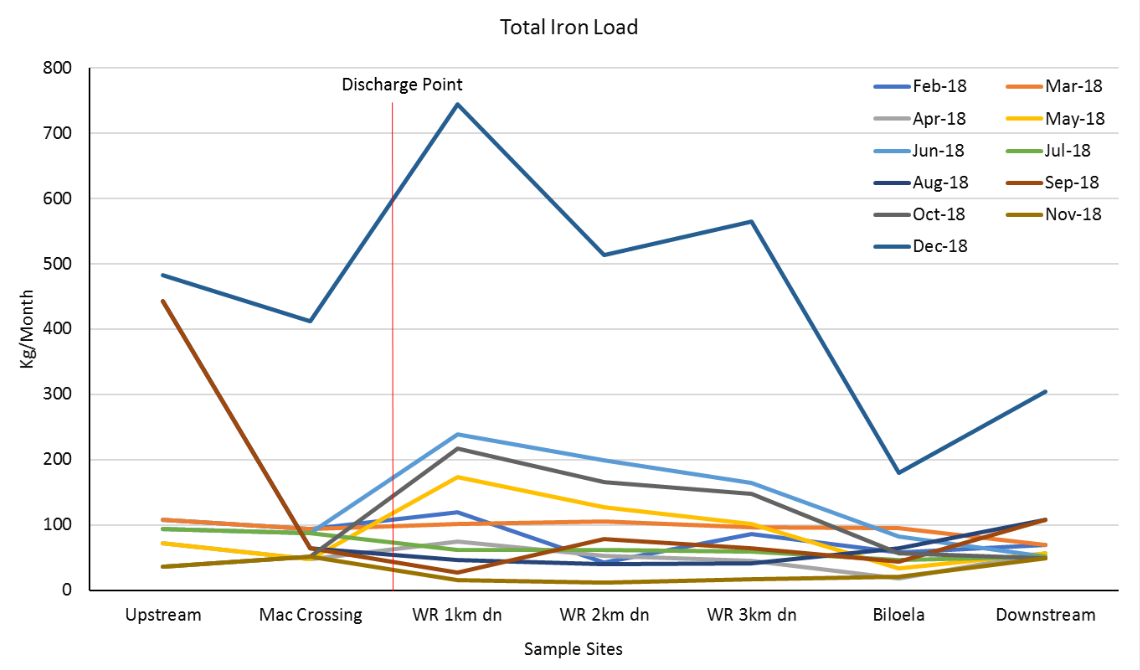
Graph 6.12 Total Nickel Load in the Wingecarribee River at all Sites



Graph 6.13 Total Zinc Load in the Wingecarribee River at all Sites

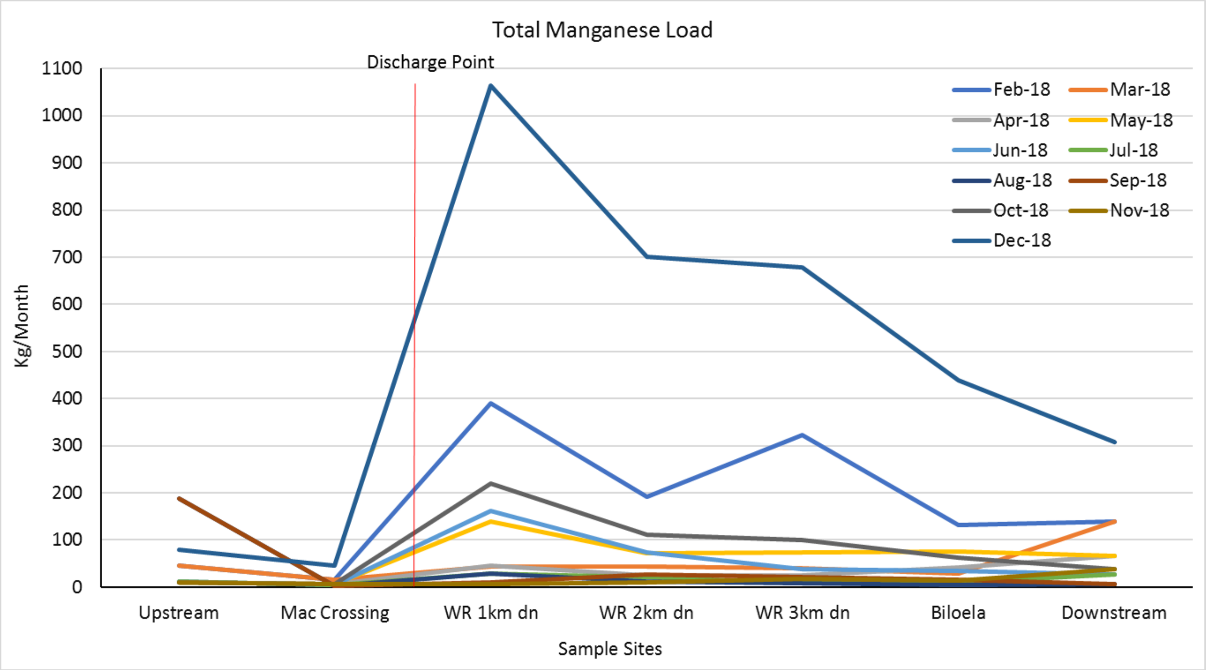
The annual snapshot of 2018 shows that Iron concentrations remain relatively low, below 200 kg/Month for most of the year. In Graph 6.14 below, December 2018 shows drastically higher Iron loads than all other months, with values up to four times higher at some sites. This is a result of much higher river flow which increased sediment load along the entire length of the Wingecarribee River within the study area.

A high concentration of 442 kg/Month was observed Upstream during September 2018. The months of May, June and October have greatest loads and they also follow the same trend, showing increased Iron levels at WR 1km dn, before declining back to levels equivalent to those Upstream. This trend is not present for the remaining months, which show low levels of Iron at all sampling sites. The principle factor governing sediment load within the river is flow.



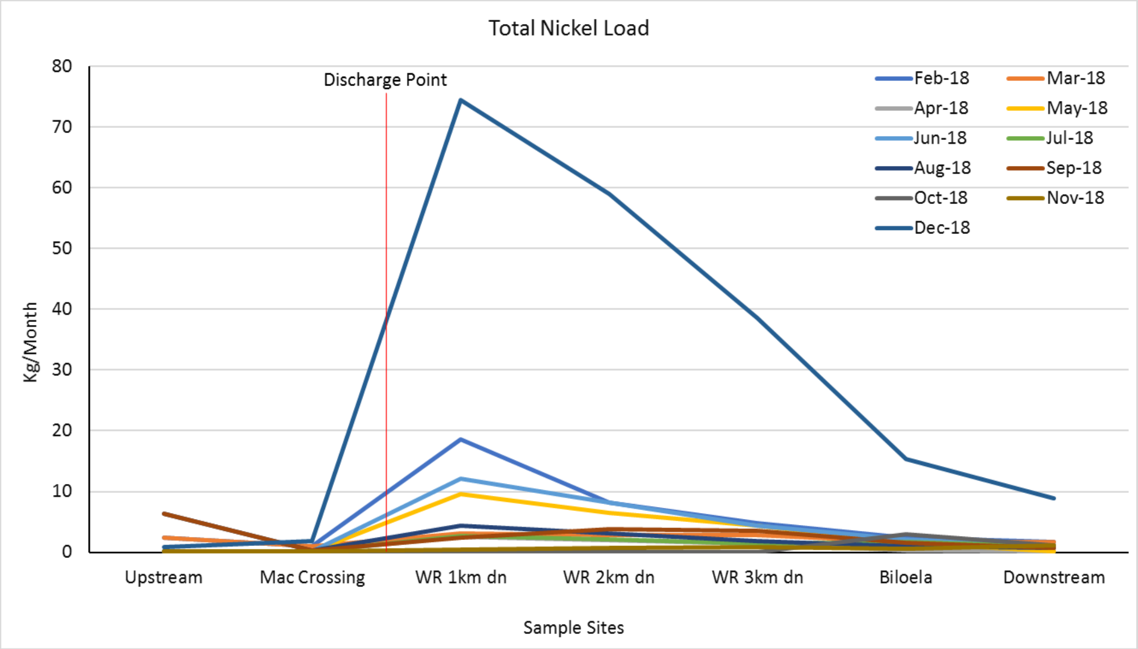
Graph 6.14 Total Iron Load in the Wingecarribee River During 2018

The total Manganese load in December was up to three times higher than in other months, with levels increasing to 1065 kg/Month at WR 1km dn due to very high river flow. Months including February, May, June and October also followed this trend, with an increased monthly load at WR 1km dn, followed by a decrease back to levels similar to upstream at Biloela. Graph 6.15 below shows that all other months have low Manganese loads, averaging 118 kg/Month over all sites.

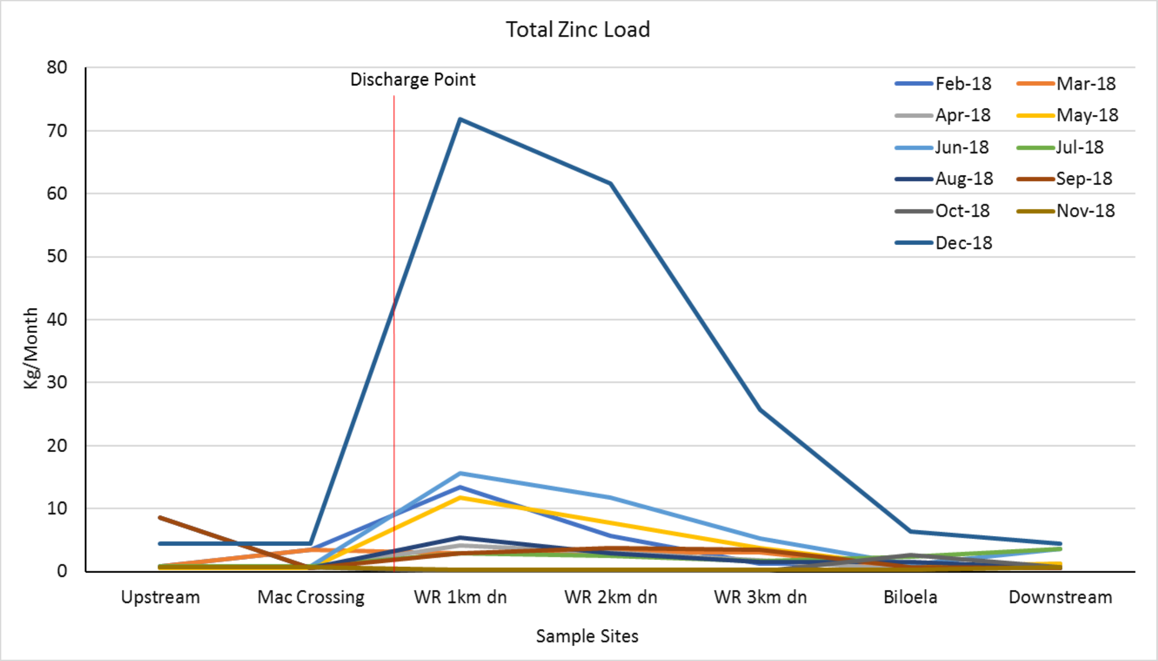


Graph 6.15 Total Manganese Load in the Wingecarribee River During 2018

Monthly loads for Nickel and Zinc are shown below in Graph 6.16 and 6.17. As observed for Iron and Manganese, December concentrations were significantly higher than all other months due to high river flow. The same trend is also observed for Nickel and Zinc, as greatest concentrations are recorded at WR 1km dn, with a decline further downstream. February, June and May display similar trends to December loads, although at levels below 20 kg/Month, for both Nickel and Zinc. All other months remain constantly low, with averages of Nickel and Zinc of 5.3 kg/Month and 5.9 kg/Month respectively.



Graph 6.16 Total Nickel Load in the Wingecarribee River During 2018



Graph 6.17 Total Zinc Load in the Wingecarribee River During 2018

Table 6.1 Iron (Total)

Date	Vol water ML for period	Initial sediment load, 400 D12	Sediment Retention before 400 C8	Remaining load at 400 C8	Sediment retained prior to Discharge	Discharge load	Total load retained in mine (Kg)
15/02/2018 - 21/02/2018	19.23	159.22	154.61	4.62	-297.30	301.91	
22/02/2018 - 14/03/2018	48.15	449.72	447.31	2.41	-313.46	315.86	
15/03/2018 - 25/04/2018	73.49	700.36	696.69	3.67	-491.65	495.32	
26/04/2018 - 2/05/2018	13.30	214.13	213.47	0.67	-52.27	52.93	
3/05/2018 - 13/05/2018	34.98	538.69	536.94	1.75	-103.89	105.64	
14/05/2018 - 30/05/2018	57.76	854.85	851.96	2.89	-102.24	105.12	
31/05/2018 - 27/06/2018	57.99	1072.82	1069.92	2.90	-189.63	192.53	
28/06/2018 - 25/07/2018	49.70	964.18	959.71	4.47	-138.66	143.14	
26/07/2018 - 27/08/2018	123.55	2347.45	2341.27	6.18	-113.67	119.84	
28/08/2018 - 24/09/2018	123.02	1869.90	1863.75	6.15	-141.47	147.62	
25/09/2018 - 29/10/2018	75.99	1565.39	1561.59	3.80	-301.68	305.48	
30/10/2018 - 26/11/2018	52.26	600.99	598.38	2.61	-93.55	96.16	
Total to date (Kg)	710.19	11337.71	11295.60	42.11	-2339.45	2381.56	8956.15

Table 6.2 Manganese (Total)

Date	Vol water ML for period	Initial sediment load, 400 D12	Sediment Retention before 400 C8	Remaining load at 400 C8	Sediment retained prior to Discharge	Discharge load	Total load retained in mine (Kg)
15/02/2018 - 21/02/2018	19.23	97.11	-1.35	98.46	-83.65	182.11	
22/02/2018 - 14/03/2018	48.15	246.53	63.56	182.97	-187.30	370.27	
15/03/2018 - 25/04/2018	73.49	238.84	13.96	224.88	-340.26	565.14	
26/04/2018 - 2/05/2018	13.30	110.12	41.76	68.36	-10.64	79.00	
3/05/2018 - 13/05/2018	34.98	99.48	-70.17	169.65	-15.39	185.04	
14/05/2018 - 30/05/2018	57.76	367.35	114.36	252.99	-5.78	258.76	
31/05/2018 - 27/06/2018	57.99	423.33	248.78	174.55	-118.88	293.43	
28/06/2018 - 25/07/2018	49.70	381.70	278.82	102.88	-98.90	201.78	
26/07/2018 - 27/08/2018	123.55	1006.93	743.77	263.16	4.94	258.22	
28/08/2018 - 24/09/2018	123.02	885.74	605.26	280.49	-45.52	326.00	
25/09/2018 - 29/10/2018	75.99	715.83	542.57	173.26	-180.86	354.11	
30/10/2018 - 26/11/2018	52.26	434.80	433.91	0.89	-98.93	99.82	
Total to date (Kg)	729.42	5007.77	3015.24	1992.53	-1181.16	3173.69	1834.08

Table 6.3 Nickel (Total)

Date	Vol water ML for period	Initial sediment load, 400 D12	Sediment Retention before 400 C8	Remaining load at 400 C8	Sediment retained prior to Discharge	Discharge load	Total load retained in mine (Kg)
15/02/2018 - 21/02/2018	19.23	4.37	-0.21	4.58	-3.60	8.17	
22/02/2018 - 14/03/2018	48.15	12.04	3.90	8.14	-8.04	16.18	
15/03/2018 - 25/04/2018	73.49	13.52	2.94	10.58	-9.70	20.28	
26/04/2018 - 2/05/2018	13.30	3.56	1.50	2.06	-1.16	3.22	
3/05/2018 - 13/05/2018	34.98	3.59	-1.90	5.49	-1.05	6.54	
14/05/2018 - 30/05/2018	57.76	13.34	3.64	9.70	-0.75	10.45	
31/05/2018 - 27/06/2018	57.99	15.38	7.15	8.23	-3.60	11.83	
28/06/2018 - 25/07/2018	49.70	12.92	5.37	7.55	-1.39	8.95	
26/07/2018 - 27/08/2018	123.55	33.61	10.87	22.73	5.31	17.42	
28/08/2018 - 24/09/2018	123.02	29.28	7.26	22.02	3.20	18.82	
25/09/2018 - 29/10/2018	75.99	23.10	9.12	13.98	-2.66	16.64	
30/10/2018 - 26/11/2018	52.26	12.70	12.33	0.37	-5.96	6.32	
Total to date (Kg)	729.42	177.40	61.96	115.44	-29.39	144.83	32.57

Table 6.4 Zinc (Total)

Date	Vol water ML for period	Initial sediment load, 400 D12	Sediment Retention before 400 C8	Remaining load at 400 C8	Sediment retained prior to Discharge	Discharge load	Total load retained in mine (Kg)
15/02/2018 - 21/02/2018	19.23	13.46	-0.17	13.63	-1.63	15.27	
22/02/2018 - 14/03/2018	48.15	37.17	15.46	21.72	-14.83	36.55	
15/03/2018 - 25/04/2018	73.49	41.23	18.59	22.63	-12.13	34.76	
26/04/2018 - 2/05/2018	13.30	4.60	2.50	2.10	-2.65	4.75	
3/05/2018 - 13/05/2018	34.98	4.40	-13.65	18.05	9.34	8.71	
14/05/2018 - 30/05/2018	57.76	15.65	6.87	8.78	-5.14	13.92	
31/05/2018 - 27/06/2018	57.99	18.02	10.08	7.94	-8.06	16.01	
28/06/2018 - 25/07/2018	49.70	17.84	10.74	7.11	-4.13	11.23	
26/07/2018 - 27/08/2018	123.55	46.21	24.22	21.99	-0.25	22.24	
28/08/2018 - 24/09/2018	123.02	37.03	15.62	21.41	1.23	20.18	
25/09/2018 - 29/10/2018	75.99	33.82	20.59	13.22	-5.17	18.39	
30/10/2018 - 26/11/2018	52.26	17.98	17.04	0.94	-11.76	12.70	
Total to date (Kg)	729.42	287.42	127.89	159.53	-55.17	214.69	72.72

6.4 Discussion

6.4.1 Ambient Sediment Quality

Generally, there are higher concentrations of Iron, Manganese, Nickel and Zinc in the river sediments upstream of the discharge point compared with the downstream sites. It can also be noted that the higher levels upstream occur once the river passes into the Hawkesbury Sandstone geological sequence at Macarthur's Crossing. The levels at the Upstream site are generally not as elevated but are still higher than the majority of samples taken at the two downstream sites.

Trends show that concentrations of all four metals tested have remained relatively constant from 2013 to 2018. These two years were both very dry, contributing to low river flow which can lead to greater deposition of metals. The increase in sediment load when the mine commenced free draining can be detected but the overall impact is minor compared to naturally occurring mineral levels at the upstream Macarthur's Crossing site.

The results show that the minerals discharged from the mine did not having an influence on sediment concentration over a broad area of the Wingecarribee River in 2013 but could be detected downstream of the discharge point in 2018. The results also suggest that the higher concentrations occur naturally and are a result of geological factors.

6.4.2 Localised Sediment Quality

There was a higher deposition of all metal precipitants at the sites WR ~300m dn and WR 1km dn. 2018 was a dry year which resulted in low river flow, allowing a build-up of mineral deposition just below the Discharge Point. Although this was similar to the data from 2013, the levels in 2018 were higher. This is likely the result of increased mineral concentration within the discharge water between the commencement of the mine free draining and the implementation of the underground treatment system. It is therefore likely that the localised sediment load was lower in 2018 than it would have been in 2016 to 2017, although no data exists to confirm this.

The data also demonstrates the influence of geology on sediment load. With generally higher concentrations upstream, it is logical to assume that this contributes to the sediment load below the discharge point, particularly during high flow events.

6.4.3 River Load

The data indicates that river load is highly dependent on river flow. However, it also shows that sediment deposited immediately below the discharge point has not yet moved further downstream. This is likely the result of the prevailing drought conditions and the lack of flushing events. The data correlates with visual evidence of mineral deposition within the immediate mixing zone, but equally corresponds to visible iron staining at Macarthur's Crossing. The difference being that the high metal bearing outcrops of Upper Hawkesbury Sandstone at Macarthur's Crossing is natural whereas the mineral deposits within the mixing zone are not.

The highest river load occurred in December 2018 after a modest increase in rain but also corresponded to the period of water transfer from the Wingecarribee River by WaterNSW.
The river load increased by a factor of up to four times higher than other months. December had an average flow of 89 ML/Day, compared to a range of 2 ML/Day to 32 ML/Day for all other months of 2018.

Low rainfall and subsequent river flow have been experienced within the Wingecarribee River for the last few years. Although this can be identified as a cause for a build-up in sediments within the mixing zone, the more obvious cause was the increase in discharge concentration of minerals.

Given the prevailing drought conditions, the river load at Biloela and the downstream site have been lower than upstream sites. This is generally considered a result of precipitation of the minerals within the mixing zone which have yet to be transported downstream.

6.5 Conclusion

From this study, it can be concluded that river loads are greater at shorter distances downstream of the discharge point, however concentrations quickly return to levels similar to upstream sites. This shows that the impact of the mine discharge still remains within the mixing zone.

The December 2018 results indicate the primary cause of high river load is flow. These deposited minerals will eventually move further downstream and will eventually dissipate within the natural sediments. The discharged minerals are essentially the same as the prevailing sandstone geology and the concentrations found within the mixing zone are still below that found at Macarthur’s Crossing.

7. Conclusion

This study represents the culmination of the investigations undertaken in 2018. It includes a comparison with similar studies undertaken in 2012. The data indicates the following:

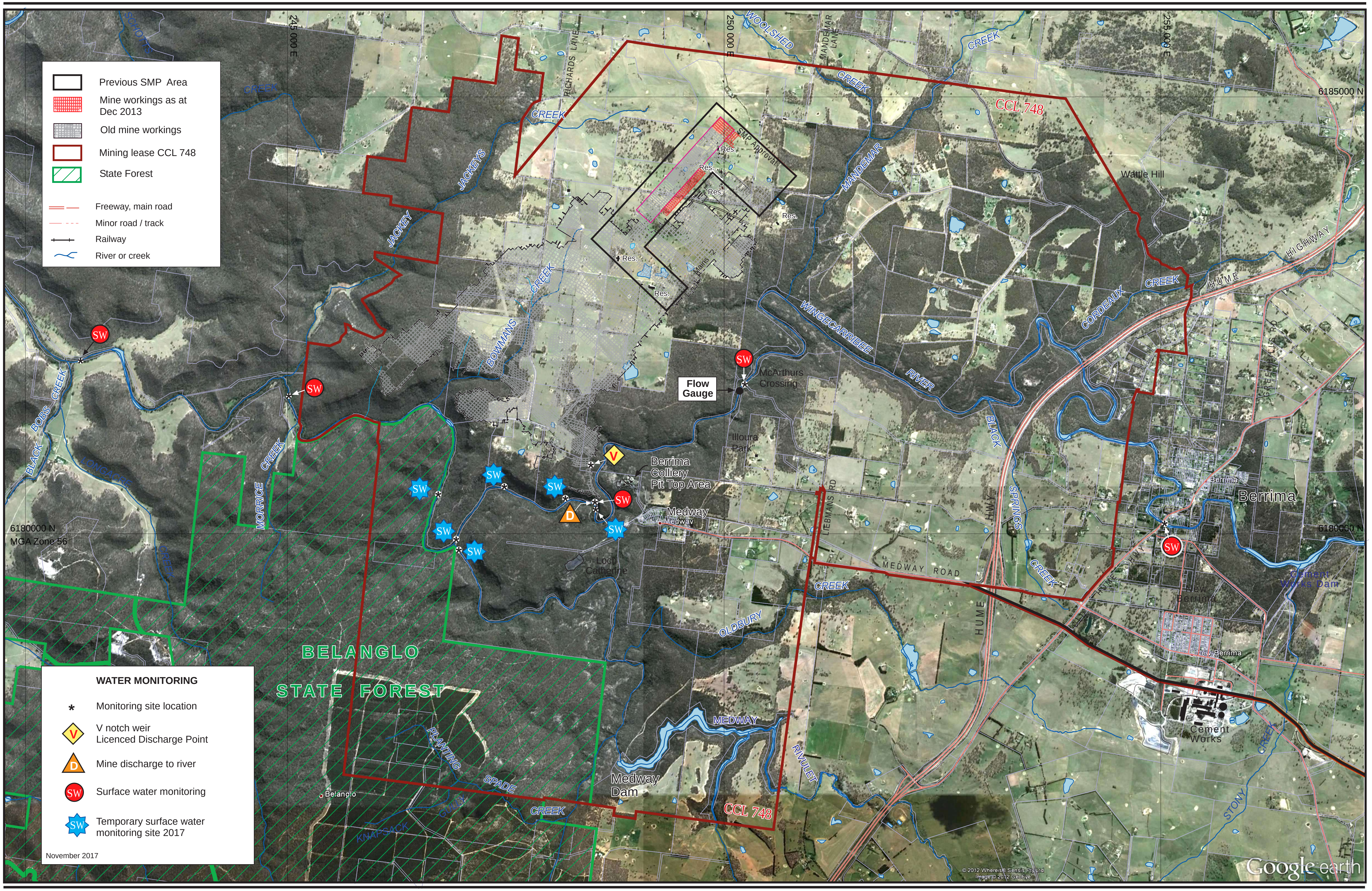
- Water quality discharged from the mine progressively improved during 2018 with the implementation of the underground water treatment system.
- Discharge quality in December 2018 and January 2019 were very similar to the long term average water quality discharged from the mine.
- The quality of water at the Biloela Site meets ANZECC guidelines for 95% ecosystem protection and recreational water.
- The ecology of the river in 2018 is comparable to the time when the mine was operational however studies undertaken in 2017 prior to the implementation of underground water treatment system showed a significant reduction in ecological health within the mixing zone.
- The ecotoxicological testing showed that the water within the mixing zone has improved between 2017 and 2018 but is below the levels found in 2013.
- Given the prevailing drought conditions, the build up in minerals within the mixing zone have not yet moved further downstream.
- River load is highly dependent on flow however some sediment movement occurred in late 2018, which was probably more related to water releases from the Wingecarribee Reservoir than natural flow.

Data collected in 2019 will be presented in February 2020. This data will provide more information on trends and potential residual impacts.

R Byrnes

Robert Byrnes
Director
International Environmental Consultants Pty Limited

Appendix A - Plans



0 1000 2000 m

N

FIGURE 1
Berrima Colliery
Water Monitoring Sites

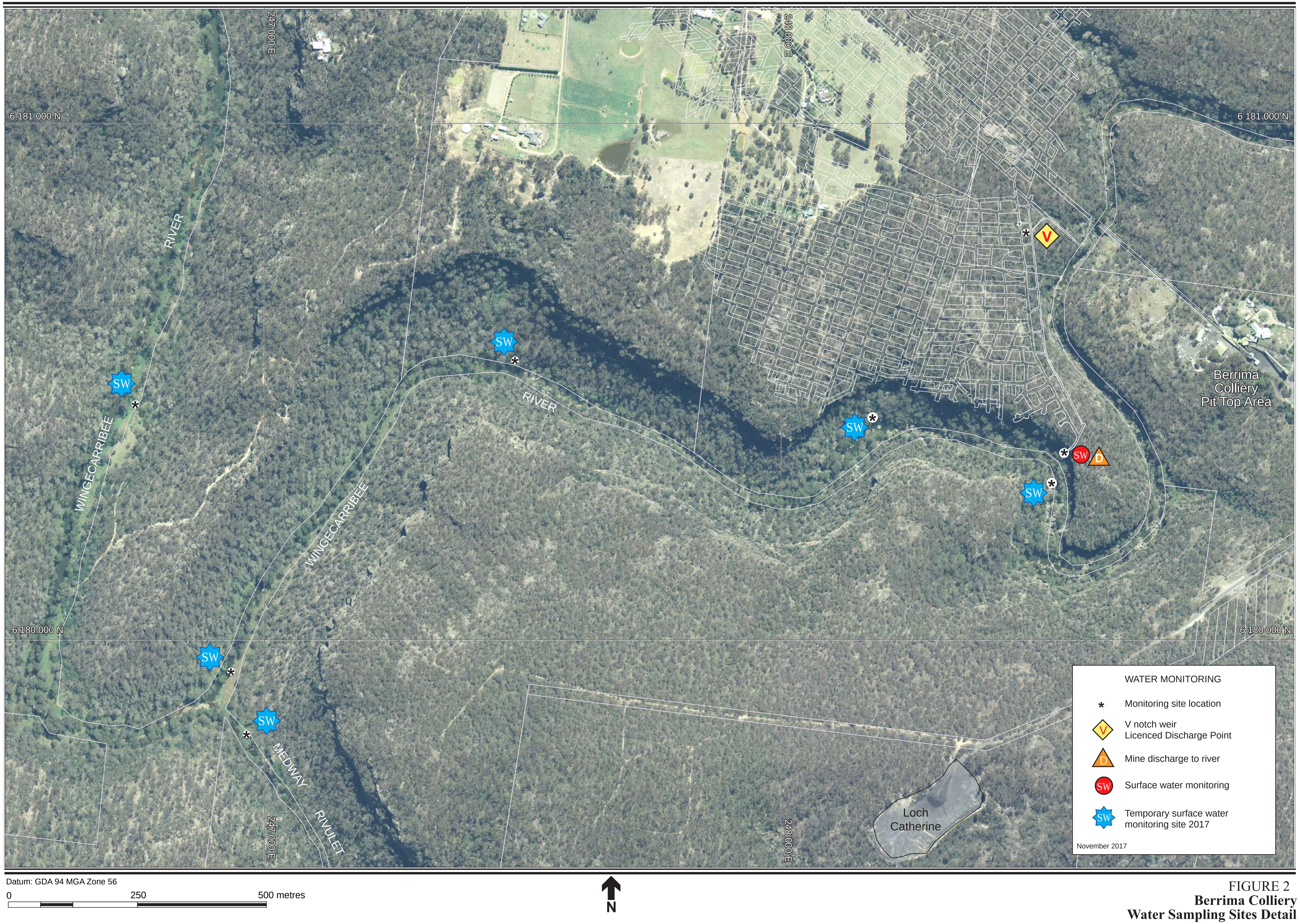


FIGURE 2
Berrima Colliery
Water Sampling Sites Detail

Appendix B – References

Altenburger, R. (2011) "Chapter 1. Understanding combined effects for metal co-exposure in ecotoxicology". *Metal ions in toxicology: effects, interactions, interdependencies*, pp.1–26

Alvial I.E., Tapia D.H., Castro M.J., Duran B.C., Verdugo C.A. (2012) Analysis of benthic macroinvertebrates and biotic indices to evaluate water quality in rivers impacted by mining activities in northern Chile, *Knowl. Managt. Aquatic Ecosyst,* 407: 01

Assessment of Ecotoxicity (2014) Chapter 7: A Framework to Guide Selection of Chemical Alternatives, *The National Academies Press*, p. 92

Basset A., Pinna M., Renzi M., (2017) Do Mining Activities Significantly Affect Feeding Behavior of Freshwater Benthic macroinvertebrates? A Case Study in South Sardinia (Italy) *Mine water Environ*, 36: 239-247

Baken, S., Nawara, S., Van Moorleghem, C., Smolders, E., (2014) Iron colloids reduce the bioavailability of phosphorous to the green alga *Raphidocelis subcapitata, water Resources,* 1(59): 198-206

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

Chapman, D. (1992). Water quality assessments. Chapman and Hall, New York, NY

Chapman P. M. (2002). "Integrating toxicology and ecology: putting the "eco" into ecotoxicology". Marine Pollution Bulletin. 44 (1): 7–15

Chapman, P. M. (1989). Current approaches to developing sediment quality criteria. Environmental Toxicology and Chemistry 8(7):589-599

Dickson, K.L., Maki, A.W., Brungs, W.A., (1984) Fate and Effects of Sediment-Bound Chemicals in Aquatic Systems: Proceedings, Pergamon Press, North Texas State University

Hill M.J., Sayer C.D., Wood P.J., (2016) When is the best time to sample aquatic macroinvertebrates in ponds for biodiversity assessment?, En*viron Monit Assess*., 188: 194

Loayza-Muro R.A., Elías-Letts R., Marticorena-Ruiz J.K. (2010) Metal-induced shifts in benthic macroinvertebrate community composition in Andean high altitude streams. Environ Toxicol Chem 29: 2761–2768

Peake, B.M., Tremblay, L.A., (2016) “Impact of Pharmaceuticals on the Environment”, *The Life-Cycle of Pharmaceuticals in the Environment`*

Ravengai S., Love D., Love I., Gratwicke B., Mandingasia O., Owen R.J.S. (2005) Impact of iron duke pyrite mine on water chemistry and aquatic life – Mazowe Valley, Zimbabwe, Water SA, 31(2): 219-228

Relyea R.; Hoverman J. (2006). "Assessing the ecology in ecotoxicology: a review and synthesis in freshwater systems". Ecology Letters. 9: 1157–1171

Rodrigues S.G. and Bueno A.A.P., (2016) Impact of iron ore mining on benthic macroinvertebrate community, R. bras. Bioci., Porto Alegre, 14(1): 15-19

Segner, H. (2011) “Reproductive and developmental toxicity in Fishes*”, Reproductive and Developmental Toxicology*

Smolders A. J. P., Lock R. A. C., Van der Velde G., Medina Hoyos R. I., Roelofs J. G. M., (2003) Effects of Mining Activities on Heavy Metal Concentrations in Water, Sediment, and Macroinvertebrates in Different Reaches of the Pilcomayo River, South America, *Archives of Environmental Contamination and Toxicology,* 44(3): 314-23

Stumpf S., Valentine-Darby P. & Gwilliam E., (2009) Aquatic Macroinvertebrates - Habitat and Life History, *NPS Inventory and Monitoring Program,* https://www.nps.gov/articles/aquatic-macroinvertebrates-habitat.htm

Sullivan R., Wright I.A., Renshaw A., Wilks M., (2014). The assessment of impacts from mining wastes on water quality and aquatic ecosystems using freshwater macroinvertebrate communities and novel bio-assay test**s**, in Vietz, G; Rutherfurd, I.D, and Hughes, R. (editors), *Proceedings of the 7th Australian Stream Management Conference*. Townsville, Queensland, Pages 369-376

Qu X., Wu N., Tang T., Cai Q., Park Y., (2010) Effects of heavy metals on benthic macroinvertebrate communities in high mountain streams, Annales de Limnologie - International Journal of Limnology, 46(4): 291-302

Water quality and macroinvertebrates, (2001) Water facts, *Water and Rivers Commission October,* Waterwatch WA, Edn.2

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