API AQUILA: BASELINE AQUATIC ECOSYSTEM SURVEYS

prepared for

Australian Premium Iron

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Study Team

Project Management: Andrew Storey
Field Reconnaissance: Andrew Storey, Adam Harman
Field work: Adam Harman, Caroline Lever, Jess Lynas & Carlin Bowyer
Macroinvertebrate identification: Adam Harman & Jess Lynas
Microinvertebrate identification: Russ Shiel (University of Adelaide)
Report: Adam Harman & Jess Lynas
Reviewed by: Andrew Storey

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Disclaimer

This document has been printed on 'Reflex Green Recycled Paper'.

Frontispiece (clockwise from left): WPIOP reference waterhole (REF60); spangled perch *Leiopotherapon unicolor*; and a WPIOP potential exposed site (KB58).

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EXECUTIVE SUMMARY

Background to the project

API joint venture plan to develop the Kens Bore resource located at the West Pilbara Iron Ore project (WPIOP). This project is based around the old WPIOP Outstation, adjacent to the Cane River, and approximately 35 km south of the Red Hill homestead. The current mine plan will require de-watering of the proposed pit area, with potential need to discharge excess water into a nearby creekline (possibly Red Hill Creek). As a result of this dewatering/discharge program, there are potential impacts to the aquatic ecosystem by dewatering drawdown of existing aquatic habitats, and dewatering discharge resulting in changes in flow regime and hydroperiod. Therefore, a comprehensive aquatic monitoring program was undertaken to document the current ecological condition of the creek, establish pre-development baseline conditions, commence data collection to allow establishment of local water quality trigger values, and assess any future changes over time associated with any dewatering/discharge operations.

Methods

Sample sites were located within areas considered to be 'potential exposed' and also sites located well away from any possible future impact (reference sites). The potential exposed sites comprised pools in Red Hill Creek, including a permanent pool immediately to the east of the ore body, pools adjacent to the south side of the ore body along the creekline, and pools downstream (to the west) of the ore body. The reference sites consisted of three pools on small creeks emerging from the ranges to the south-east of the project area, and, two pools on the creekline further to the north, adjacent to the Cochrane and Jewel ore bodies.

Aquatic ecosystem sampling of the WPIOP sites involved *in situ* water quality measurements as well as water sample collection for analysis of nutrients, ions and metals. Microinvertebrates and macroinvertebrates were sampled using 53 μm and 250 μm mesh dip-nets, respectively. Hyporheic fauna samples were collected from each site by digging a hole in alluvial gravels in dry streambed adjacent to the waters edge. The hole was allowed to fill with water while other aquatic sampling was undertaken, and then swept with a modified 53 μm mesh plankton net. Fish were sampled using a variety of methods, including seine nets, gill nets and dip nets. Sampling was undertaken in the dry season of 2008 (December) and wet season of 2009 (April).

Results

The water quality of WPIOP waterbodies was generally good and characterised by circum-neutral pH, high dissolved oxygen levels and fresh waters. However, the dissolved oxygen concentration recorded from KB50 in the dry season was extremely low and the fish fauna were showing obvious signs of stress at the time of sampling (air-breathing). The total nitrogen levels recorded from most sites in the study area were elevated and in excess of the ANZECC/ARMCANZ (2000) trigger value for the protection of aquatic systems. In addition, a number of heavy metals appear to be naturally elevated, including boron, copper and zinc. Multivariate statistics showed no significant difference in water quality between potential exposed and reference sites.

The microinvertebrate fauna from WPIOP waterbodies was highly diverse with 135 taxa being collected across the dry and wet seasons. A significantly greater number of taxa was recorded in the wet season, but there was no difference in taxa richness between potential exposed and reference sites. Of interest within the microinvertebrate fauna was the collection of a number of taxa endemic to Australia, including the protist *Difflugia australis*, the rotifers *Lecane noobijupi* and *Lecane batillifer*, and the Cladocera *Alona rigidicaudis* and *Moina* cf. *micrura*. Australian endemic species were recorded in low proportions from both potential exposed and reference sites.

Furthermore, two microinvertebrate species collected during the current study were classified as new records for either Australia or Western Australia. The collection of the protist *Difflugia capreolata* from the potential exposed site KBNW2 was a new record for Australia. The rotifer *Cephalodella gigantea* collected from the reference site JC91 constituted a new record for Western Australia.

Microinvertebrate assemblages were found to be significantly different between type (potential exposed vs reference sites). The wet season microinvertebrate assemblages of the two Jewel Cochrane sites (reference sites) were more similar to the wet season Kens Bore sites (potential exposed). There was no significant difference in microinvertebrate assemblages between season.

Given that microinvertebrate assemblages are different between potential exposed and reference sites prior to the commencement of mining, changes in Bray-Curtis similarities can be used in future monitoring to assess any effect of dewatering and possible discharge operations.

Three of the reference sites were not able to be sampled for hyporheic fauna due to the lack of appropriate substrate (gravel) and presence of rocky outcrops (REF59, REF60 and REF61).

Sampling of hyporheic habitats from seven sites in the WPIOP study area in the dry season of 2008, and nine sites in the wet season 2009 revealed the presence of hyporheic fauna. The majority of taxa collected in hyporheic samples however, were classified as stygoxene (69%), i.e. they do not have specialised adaptations for groundwater habitats and were likely surface forms present in samples. Of the 45 taxa collected, 2% were classified as occasional hyporheos stygophiles, 2% were considered true stygobites, and 5% were considered possible hyporheic fauna. None were considered to be permanent hyporheos stygophiles. Species considered to be hyporheos were the stygobitic amphipod ?Paramelitidae spp., occasional stygophile copepod *Microcyclops varicans*, and possible hyporheic taxa Nematoda spp., and Oligochaeta spp. Hyporheic fauna were collected from all sites in both the wet and dry seasons.

Stygobitic amphipods were recorded from a number of sites in the WPIOP study area, including the potential exposed sites KB57, KB58, and KBNW2, and reference site JC91. Although these specimens are still awaiting formal identification they were known to be restricted to groundwater and hyporheic environments because they exhibited a number of characteristics unique to stygofauna, including a lack of pigmentation, eyes reduced or absent, and elongate appendages. These amphipods may be closely related, or the same species as stygobitic amphipods recorded from other parts of the Pilbara (WRM unpub. data). Genetic analysis of the DNA of these specimens is required to determine their conservation significance, and whether they represent a range extension of a currently known species, or a new species to science.

A total of 128 taxa of macroinvertebrates were recorded from the twenty sites sampled (dry and wet seasons). The composition of macroinvertebrate taxa was typical of freshwater systems throughout the world (Hynes 1970), and was dominated by Insecta (84% of taxa). Although the mean number of macroinvertebrate taxa collected was slightly higher in the wet season, there was no significant difference in taxa richness between seasons. Nor was there any significant difference in the number of taxa collected between site type (i.e. potential exposed vs reference sites).

The majority of macroinvertebrate taxa recorded were common, ubiquitous species. Of interest however, was the collection of species known only from the Pilbara region of Western Australia. Pilbara endemic species included the Odonate *Nannophlebia injibandi* and the haliplid beetle *Haliplus pilbaraensis*. Both species were collected from the potential exposed site KB58.

Multivariate statistics showed that there was no significant difference between macroinvertebrate assemblages of potential exposed sites vs reference sites. Generally, the within type variation was high, i.e. macroinvertebrate assemblages were highly variable among potential exposed sites, and highly variable amongst reference sites. This is a good result in terms of long-term monitoring as it will better enable testing for any effects of mining. Because the macroinvertebrate assemblages of all sites are currently similar / indistinguishable, any future separation of exposed sites from reference sites would signal some effect.

There was a significant difference in macroinvertebrate assemblages between season.

Five of the twelve freshwater fish species known from the Pilbara were recorded during the current study. These were the western rainbowfish *Melanotaenia australis*, Hyrtl's tandan (eel-tailed catfish) *Neosiluris hyrtlii*, spangled perch *Leiopotherapon unicolor*, Fortescue grunter *Leiopotherapon aheneus* and barred grunter *Amniataba percoides*. Spangled perch and western rainbowfish were the most common species recorded. Spangled perch were recorded from all sites, while the barred grunter was only recorded from JC91.

Generally, the fish recorded from the WPIOP study area are common widespread species. However, the Fortescue grunter has a restricted distribution within the Pilbara Region of Western Australia. It is only known from the Fortescue, Robe and upper Ashburton (Nicholl's Spring) river systems (Allen *et al.* 2002). The Fortescue grunter is reasonably common within its range. This species is currently listed as 'Lower Risk Near Threatened' on the IUCN Redlist of Threatened Species (IUCN 2009). Its status is considered to require updating (IUCN 2009).

A number of recommendations are provided for future monitoring.

1 INTRODUCTION

1.1 Background

The West Pilbara Iron Ore Project (WPIOP) is located approximately 200 km south-south-west of Karratha. Exploration has been based around the old WPIOP Outstation, adjacent to the Cane River, and approximately 35 km south of the Red Hill homestead. Kens Bore is the largest of the ore bodies, stretching 10 km east to west. The resource area lies within the floodplain of two significant creeks, Red Hill Creek and Kunada Creek. Red Hill Creek and its surrounds is a significant tributary of the Robe River in the West Kimberley region of Western Australia. Red Hill Creek has a catchment of approximately 300 km² upstream from the deposit. Drainage is from the southern tip of Kens bore and then flows northwest, where it meets Kunada Creek, until it joins the Robe River some 40 km away (Aquaterra 2008).

The WPIOP mine plan indicates that de-watering of the developing pit area at Kens Bore will be necessary, with potential need to discharge excess water into a creekline (possibly Red Hill Creek). As a result of this dewatering/discharge program, there are potential impacts to the aquatic ecosystem. This may be due to dewatering drawdown and drying of existing aquatic habitats, and discharge of dewatering water, resulting in changes in flow regime and hydroperiod. Therefore, a comprehensive aquatic monitoring program was undertaken to document the current ecological condition of the creek, establish pre-development baseline conditions, commence data collection to allow establishment of local water quality trigger values, and assess any future changes over time associated with any dewatering/discharge operations.

This report presents the findings of the first two rounds of sampling at locations throughout the study area (December 2008 / April 2009).

1.2 Study objectives

The purpose of this project was to establish baseline conditions for future assessment of effects, if any, of development and operation of the project on the ecological health of Red Hill Creek and associated pools. Data collected will identify ecological values and conservation significance of the aquatic ecosystems in the immediate vicinity of the resource area, allow future impact assessment and will allow monitoring of changes in water quality and aquatic fauna over the life of the project. ANZECC/ARMCANZ (2000) recommend at least three years baseline data are required to establish local trigger levels for water quality, with similar data requirements for assessing changes in aquatic fauna. The report has been prepared at the request of the WPIOP to assist in project design consideration and establish a pre-development baseline dataset.

2 METHODS

2.1 Study area

The WPIOP study area is located within the Pilbara region of Western Australia (see Figure 1). The Pilbara is situated in the north west of Western Australia, extending across an area of 507 896 km^2, from the Indian Ocean to the Northern Territory border (and includes off-shore islands). The Pilbara is a mineral rich region that is thought to be around 2.8 billion years old.

Climate of the region is semi arid and is characterised by high summer and medium winter temperatures, low and variable rainfall and high evaporation. Most rainfall occurs during the summer months and is associated with cyclonic and monsoonal events; when flooding frequently occurs along creeks and rivers (Gardiner 2003). Due to the nature of cyclonic events and thunderstorms, total annual rainfall in the region is highly unpredictable and individual storms can contribute several hundred millimetres of rain at one time. Average annual pan evaporation in the Pilbara is ten times greater than rainfall. Temperature ranges are generally greater in inland areas, away from the moderating effects of onshore winds common to the coastal districts.

Streamflow in Pilbara river systems reflects rainfall patterns and is highly seasonal and variable. Flows occur as a direct response to rainfall, with peak flows tending to occur within 24 hours of a rainfall event and continuing for several days (Hamersley Iron Pty Ltd 1995).

Due to the aridity of the region, permanent waterbodies are rare, with permanent surface water being "restricted to springs and some permanent groundwater-fed pools in the beds of large rivers" (Halse *et al.* 2002). Predictable sources of water would likely support richer aquatic faunas than ephemeral systems (Kay *et al.* 1999). Permanent pools and flowing reaches of spring-fed systems would provide vital refuges for aquatic fauna, including fish and macroinvertebrates, particularly in the dry season and over dry years. Halse *et al.* (2002) suggested that such systems provide an important "source of animals for colonisation of newly flooded pools and maintenance of populations of invertebrate species at the regional level". The study area is traversed by a number of creeks, including Red Hill Creek and Kunada Creek (which flow into the Robe River), headwaters of the Cane River (flows to the coast near Onslow), and Duck and Urandy Creeks (that feed into the Ashburton River). All creek lines in the area are episodically-flowing systems, but support permanent waterholes to differing degrees.

Figure 1. Location of the study area within the Pilbara region of Western Australia.

2.2 Sites and sampling design

To establish baseline conditions for future assessment of any mine effects, a number of sampling locations were selected within areas considered to be 'potential exposed' and also sites located well away from any possible future impact (reference sites). Statistical analyses rely upon obtaining replicate samples (*viz.* sites) to characterise within and between site spatial variability in the parameters being measured (*i.e.* species richness, assemblage composition, water quality parameters) and to provide statistical power to test for between 'type' differences (*i.e.* the ability to statistically detect differences/affects if they exist). Therefore, replicate sites were sampled within 'potential exposed' and 'reference' areas.

A total of 11 sites were sampled during the current study; 9 in the dry season of 2008 and 11 during the wet of 2009 (Table 1). The additional sites after the wet season reflected additional surface water downstream of the Ken's Bore project area. The potential exposed sites comprised pools in Red Hill Creek, including a permanent pool immediately to the east of the ore body, pools adjacent to the south side of the ore body along the creekline, and pools downstream (to the west) of the ore body (see Figure 2 and Table 1). The reference sites consisted of three pools on small creeks emerging from the ranges to the south-east of the project area, and, two pools on the creekline further to the north, upstream of the Cochrane and Jewel ore bodies (see Figure 2 and Table 1).

Table 1. Site details for each of the eleven sites sampled, including system, site code, type, and GPS location (WSG84). PE refers to sites which are 'potential exposed', and R = reference sites.

				GPS		Sampled in	
System	**Site**	**Site Code**	**Type**	**Easting**	**Northing**	**Dry 08**	**Wet 09**
Red Hill Creek	Kens Bore 50	KB50	PE	50 421338	7556990	✓	✓
	Kens Bore 56	KB56	PE	50 416332	7557510	✓	✓
	Kens Bore 57	KB57	PE	50 416325	7558252	✓	✓
	Kens Bore 58	KB58	PE	50 415428	7559060	✓	✓
	Kens Bore North West 1	KBNW1	PE	50 406204	7566799		✓
	Kens Bore North West 2	KBNW2	PE	50 414068	7560455		✓
Cane River	Reference 59	Ref59	R	50 422583	7552046	✓	✓
	Reference 60	Ref60	R	50 422302	7551796	✓	✓
	Reference 61	Ref61	R	50 423462	7548478	✓	✓
Mungarathoona Ck	Jewel Cochrane 90	JC90	R	50 415410	7575819	✓	✓
	Jewel Cochrane 91	JC91	R	50 414449	7576516	✓	✓

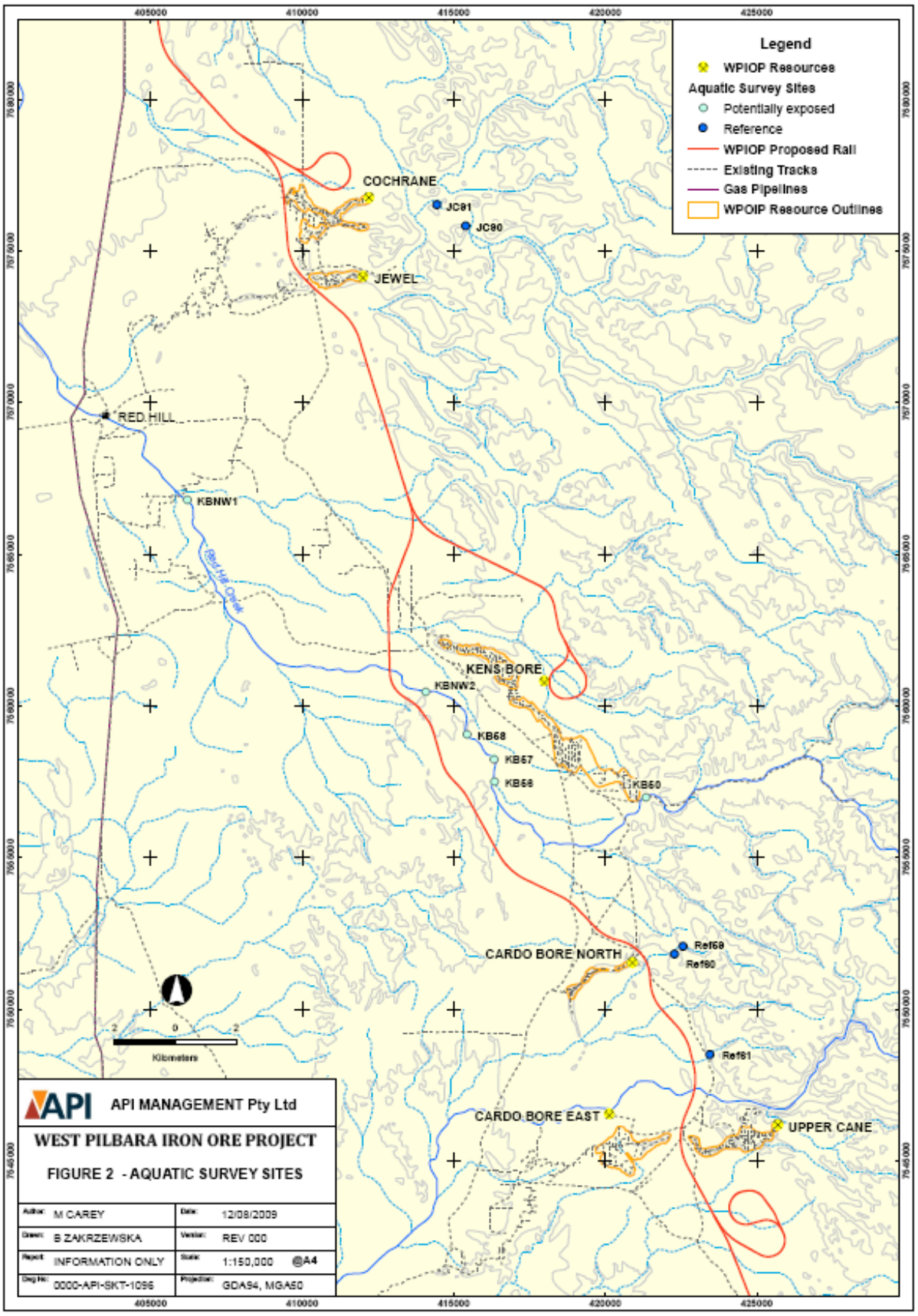


Figure 2. Map showing the location of all WPIOP sampling sites (refer Table 1 for explanation of site codes).

2.3 Water quality

At each site a number of water quality variables were recorded *in situ* using portable WTW field meters, including pH, electrical conductivity (µS/cm), dissolved oxygen (% and mg/L), and water temperature (°C). Water depth was measured using a graduated pole. Undisturbed water samples were taken for laboratory analyses of dissolved metal, ion, and nutrient concentrations. These samples were filtered through 0.45 µm Millipore nitrocellulose filters. All water samples were kept cool in an esky while in the field, and nutrient samples then frozen as soon as possible for subsequent transport to the laboratory. All laboratory analyses were conducted by the Natural Resources Chemistry Laboratory, Chemistry Centre, WA (a NATA accredited laboratory). Water quality variables measured are summarised in Table 2.

Table 2. Water quality parameters measured.

Parameter	Units	Parameter	Units
pH	pH units	Total Nitrogen (total N)	mg/L
Electrical conductivity	µS/cm	Total Phosphorus (total P)	mg/L
Dissolved oxygen	% saturation	Arsenic (As)	mg/L
Dissolved oxygen	mg/L	Cadmium (Cd)	mg/L
Redox potential	mV	Cobalt (Co)	mg/L
Water temp	°C	Chromium (Cr)	mg/L
Maximum water depth	m	Copper (Cu)	mg/L
Sodium (Na)	mg/L	Iron (Fe)	mg/L
Potassium (K)	mg/L	Manganese (Mn)	mg/L
Calcium (Ca)	mg/L	Molybdenum (Mo)	mg/L
Magnesium (Mg)	mg/L	Nickel (Ni)	mg/L
Chloride (Cl)	mg/L	Lead (Pb)	mg/L
Carbonate (CO3)	mg/L	Selenium (Se)	mg/L
Hydrogen carbonate (HCO3)	mg/L	Vanadium (V)	mg/L
Sulphate (SO_4)	mg/L	Zinc (Zn)	mg/L
Nitrate (NO_3)	mg/L		
Ammonia (NH_4)	mg/L		

Water quality data were compared against ANZECC/ARMCANZ (2000) water quality guidelines. ANZECC/ARMCANZ (2000) provides trigger values for a range of water quality parameters for the protection of aquatic ecosystems. These trigger values may be adopted in the absence of adequate site-specific data. ANZECC/ARMCANZ (2000) recommends different levels of species protection applied to different levels of ecosystem condition. The 99% value is applied to high conservation/ecological value ecosystems, the 95% value to slightly to moderately disturbed ecosystems and the 90% or 80% values to highly disturbed ecosystems. In the ANZECC/ARMCANZ (2000) water quality management framework, the decision about the ecosystem condition is typically a joint one between stakeholders. Based on the observed condition of creeks in the vicinity of Kens Bore, it is suggested that either the 99% or possibly the 95% values are applied. When applying trigger values (TVs), ANZEEC/ARMCANZ (2000) state the following:

> "*Trigger values are concentrations that, if exceeded, would indicate a potential environmental problem, and so 'trigger' a management response, e.g. further investigation and subsequent refinement of the guidelines according to local conditions*." (Section 2.1.4); and
>
> "*Exceedances of the trigger values are an 'early warning' mechanism to alert managers of a potential problem. They are not intended to be an instrument to assess 'compliance' and should not be used in this capacity*." (Section 7.4.4)

Hence, TVs should not be used in a 'pass-fail' approach to water quality management. Their main purpose is to inform managers and regulators that changes in water quality are occurring and may need to be investigated. In the case of baseline data collection, the guidelines may be used to establish background levels relative to TVs, and show where certain elements may be naturally elevated (i.e. due to geological features). This allows future discrimination of mine effects from natural enrichment. Where background levels are elevated, then it is desirable to establish site-specific TVs.

The guidelines recommend, that where an appropriate default TV does not exist, or the default TV is consistently lower than natural background concentrations, natural background data should be used to derive the TV. In these instances, the 80th percentile (and 20th percentile in the case of variables that require an upper and lower guidelines, e.g. pH) of a baseline dataset should be used. This value is then compared to the median value of the subject water (i.e. the dewatering water) (for further details see Sections 3.3.2.4 and 7.4.4 of ANZECC/ARMCANZ 2000). It is also recommended that TV are based on at least three years of monthly monitoring data.

2.4 Microinvertebrates

Microinvertebrate fauna consists of microscopic fauna including micro-crustacea (ostracods, copepods and cladocera), protists and rotifers. Microinverterbates are used as bioindicators throughout the world for many reasons. Firstly, the microinvertebrate community holds a strategic position in food webs (Bunn and Boon 1993, Zrum and Hann 1997, Bunn and Davies 1999, Jenkins and Boulton 2003). They regulate the biomass of phytoplankton in the water column and epiphyton on submerged aquatic macrophytes through grazing (Zrum and Hann 1997). They also provide a food source for other organisms, such as macroinvertebrates (Bunn and Boon 1993, Jenkins and Boulton 2003). Many waterbirds feed directly on microinvertebrate fauna (Crome 1985), and most fish species depend on them for their first feed after hatching (Geddes and Puckridge 1989). Therefore, any change in the microinvertebrate community will ultimately result in changes to the entire aquatic ecosystem. Secondly, due to their short life cycle, rapid changes occur in their populations in response to disturbance in the ecosystem (Kaur and Ansal 1996). Lastly, they have intimate contact with the surrounding environment, being planktonic, and continually exposed to the ambient water quality. Hence, they are vulnerable to environmental pollutants and provide a useful biomonitoring tool (Kaur and Ansal 1996). The microinvertebrate community also plays a role in nutrient cycling within wetland systems (Baldwin and Mitchell 2000).

Microinvertebrate samples were collected by gentle sweeping over an approximate 15 m distance with a 53 μm mesh pond net. Care was taken not to disturb the benthos (bottom sediments). Samples were preserved in 70% ethanol and sent to Dr Russ Shiel of The University of Adelaide for processing. Dr Shiel is a world authority on microfauna, with extensive experience in fauna survey and impact assessment across Australasia, including the Pilbara region of Western Australia.

Microinvertebrate samples were processed by identifying the first 200-300 individuals encountered in an agitated sample decanted into a 125 mm^2 gridded plastic tray, with the tray then scanned for additional missed taxa also taken to species, and recorded as 'present'. Specimens were identified to the lowest taxon possible, i.e. species or morphotypes. Where specific names could not be assigned, vouchers were established and specimens/images sent to various world experts. These vouchers are held by Dr Shiel at The University of Adelaide, South Australia.

2.5 Hyporheic fauna

The hyporheic zone is increasingly becoming recognised as a critical component of many streams and rivers (Edwards 1998). This zone is known to function in a number of important ways. For example, hyporheic biogeochemical processes can influence surface water quality (Bencala 1984), and streams with an extensive hyporheic zone can retain and process solutes more efficiently than those without (Valett *et al.* 1996). The hyporheic zone is also thought to provide a rearing habitat (Brunke and Gonser 1999) and important refuge for aquatic invertebrates, buffering them from floods (Palmer *et al.* 1992, Dole-Oliver & Marmonier 1992, Edwards 1998), disturbance in food supply (Edwards 1998), and drought (Cooling and Boulton 1993, Edwards 1998, Coe 2001, Hose *et al.* 2005). The often diverse and abundant fauna[1] of hyporheic zones has been found to dominate the biological productivity of rivers (Stanford and Ward 1988, Smock *et al.* 1992).

Hyporheic fauna were targeted in the current study as they may be impacted by dewatering and discharge operations. Anthropogenic alterations to hydrology (i.e. impoundments, dewatering, and discharge) impact hydraulic conductivity and/or hydraulic head patterns, thereby decreasing available hyporheic habitat and the strength of its connection to surface waters. In addition, anthropogenic processes which affect the rates of sediment input and transport can also affect the availability of hyporheic habitat (Edwards 1998), i.e. through the blocking of interstitial spaces.

In studies undertaken in the Northern Hemisphere, the largest numbers of groundwater animals have been found in shallow groundwater of the hyporheos rather than the deeper phreatic zone (Marmonier *et al.* 1993, Rouch and Danielopol 1997).

Interstitial fauna exhibit unique traits and adaptations to survive life in sediment pores. They have long, slender and flexible bodies which facilitate movement through interstitial spaces, and their small, hard, blunt bodies allow them to force their way through (Williams 1984). Some organisms are simply very small. Stygobites are blind and lack pigmentation.

Hyporheic fauna was sampled at all sites where possible. Sampling was conducted by digging a hole approximately 20 cm deep and 50 cm diameter in alluvial gravels in dry streambed adjacent to the waters edge (Plate 2). The hole was allowed to fill with water, and then the water column was swept with a modified 53 μm mesh plankton net immediately after the hole had filled, and again after approx. 30 minutes while other sampling was conducted (Plate 1). Hyporheic sampling was not possible at all sites, due to the presence of rocky outcrops (i.e. reference sites REF59, REF60 and REF61).

Plate 1. Sampling for hyporheic fauna.

Samples were preserved in 70% ethanol and returned to the laboratory for processing. Any hyporheic fauna present was removed from samples by sorting under a low power dissecting microscope. Specimens were sent to appropriate taxonomic experts for identification and confirmation of their status as hyporheic fauna.

[1] Known as hyporheic fauna, or hyporheos.

Chironomidae (non-biting midges) were sent to Dr Don Edward (The University of Western Australia), Hydracarina (aquatic mites) to Dr Mark Harvey (Western Australian Museum), Amphipoda to Dr Bill Humphreys (Western Australian Museum), Copepoda to Dr Danny Tang (The University of Western Australia), and Oligochaeta to Dr Adrian Pinder (Department of Environment and Conservation).

All taxa recorded from hyporheic samples were classified using Boulton's (2001) categories;

- stygobite – obligate groundwater species, with special adaptations to survive such conditions
- permanent hyporheos stygophiles - epigean[2] species which can occur in both surface- and groundwaters, but is a permanent inhabitant of the hyporheos
- occasional hyporheos stygophiles – use the hyporheic zone seasonally or during early life history stages
- stygoxene (species that appear rarely and apparently at random in groundwater habitats, there by accident or seeking refuge during spates or drought; not specialised for groundwater habitat).

Hyporheic fauna was not sampled for comparison with historic data, but rather to provide some baseline data for sites in the East Pilbara region.

2.6 Macroinvertebrates

Macroinvertebrates (*i.e.* fauna retained by a 250 µm aperture mesh) typically constitute the largest and most conspicuous component of aquatic invertebrate fauna in both lentic (still) and lotic (flowing) waters. Macroinvertebrates are used as a key indicator group for bioassessment of the health of Australia's streams and rivers under the National River Health Program (Schofield & Davies 1996), and have inherent value for biological monitoring of water quality (ANZECC/ARMCANZ 2000).

Plate 2. Using the 250 µm mesh pond net to selectively collect aquatic macroinvertebrates.

Sampling was conducted with a 250 µm mesh FBA pond net to selectively collect the macroinvertebrate fauna (Plate 2). All meso-habitats were sampled, including trailing riparian vegetation, woody debris, open water column and benthic sediments with the aim of maximising the number of species recorded. Each sample was washed through a 250 µm sieve to remove fine sediment, leaf litter and other debris. Samples were then preserved in 70% ethanol.

In the laboratory, macroinvertebrates were removed from samples by sorting under a low power dissecting microscope. Collected specimens were then identified to the lowest possible level (genus or species level) and enumerated to $\log_{10}$ scale abundance classes (*i.e.* 1 = 1 - 10 individuals, 2 = 11 - 100 individuals, 3 = 101-1000 individuals, 4 = >1000). In-house expertise was used to identify invertebrate taxa using available published keys and through reference to the

[2] Epigean – living or occurring on or near the surface of the ground.

established voucher collections held by WRM. External specialist taxonomic expertise was sub-contracted to assist with Chironomidae (non-biting midges) (Dr Don Edward, The University of Western Australia) and Hydracarina (aquatic mites) (Dr Mark Harvey, the Western Australia Museum).

2.7 Fish

Fish diversity has been used as an indicator of ecosystem health worldwide (Karr 1991, Oberdoff and Hughes 1992, Hugueny *et al.* 1996, An and Choi 2003). Because fish continually inhabit the receiving water, they integrate the chemical, physical and biological histories of the river. Most fish species have a long life span and therefore reflect both long-term and current water quality. Sampling fish assemblages can be used to assess a range of environmental disturbances, such as changes in habitat, water quality and land use (Hugueny *et al.* 1996). Fish also tend to be the most conspicuous biota in the freshwater systems, are relatively easy to sample and identify, and provide a significant food source for indigenous people.

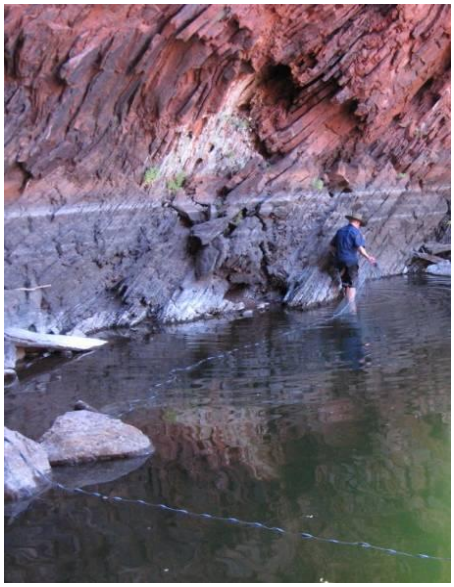


Plate 3. Gill nets set at Reference site 59.

A number of methods were used at each site to effectively collect as many species/individuals as possible. Fish sampling methods included seine, gill and dip netting.

Light-weight fine mesh gill nets (10 m net, with a 2 m drop, using both 13 mm and 19 mm stretched mesh) were used at each site and were set in deeper water for the duration of sampling at that site (Plate 3). Smaller species and juveniles were sampled by beach seine (10 m net, with a 2 m drop and 6 mm mesh) deployed in shallow areas where there was little vegetation or large woody debris. Generally, two seines were conducted at each site to maximise the number of individuals caught.

All fish were identified in the field, measured and then released alive. Fish nomenclature followed that of Allen *et al.* (2002). Fish measurements provided information on the size structure, breeding and recruitment of fish populations.

2.8 Data analysis

2.8.1 Multivariate analysis

Multivariate analyses were performed using the PRIMER package v 6 (Plymouth Routines in Multivariate Ecological Research; Clarke and Gorley 2006) to investigate differences in aquatic invertebrate assemblages among sites. The PRIMER package, developed for multivariate analysis of marine fauna samples, has been applied extensively to analysis of freshwater invertebrate data. Analyses used included:

1. Describing pattern amongst the fauna assemblage data using ordination techniques based on Bray-Curtis similarity matrices. Ordination of data was by Multi-Dimensional Scaling (MDS) (Clarke and Warwick 2001). Ordinations were depicted as two-dimensional plots based on the site by site similarity matrices.

2. For any groups found in (1) or selected *a priori* (*i.e.* treatment type – potential exposed vs reference, or season – wet vs dry), Analysis of Similarity (ANOSIM) – effectively an analogue of the univariate ANOVA – was conducted to determine if groups were significantly different from one another. The ANOSIM test statistic reflects the observed differences *between* groups with the differences amongst replicates *within* the groups. The test is based upon rank similarities between samples in the underlying Bray-Curtis similarity matrix. The analysis presents the significance of the overall test (Significance level of sample statistic), and significance of each pairwise comparison (Significance level %), with degree of separation between groups (R-statistic), where R-statistic >0.75 = groups well separated, R-statistic >0.5 = groups overlapping but clearly different, and R-statistic >0.25 = groups barely separable. A significance level <5% = significant effect/difference.
3. The SIMPER routine was used to examine which taxa were contributing to the differences of any groups that were found to be different according to the ANOSIM procedure or otherwise found to be separated in cluster or ordination analyses.
4. The relationship between the environmental and biotic data was assessed in two ways:
   - the BIOENV routine was used to calculate the minimum suite of parameters that explain the greatest percent of variation (i.e. the parameters which most strongly influence the species ordination), and
   - for visualisation, the numeric value of key environmental data (as determined by BIOENV) were superimposed onto MDS ordinations, as circles of differing sizes – so-called 'bubble plots'.

Water quality data were similarly analysed using MDS to discern patterns, gradients and similarities in water quality amongst the sites and seasons. In this case, however, the MDS was based on a Euclidean Distance measure rather than Bray-Curtis. Water quality variables which were not normally distributed were appropriately transformed and all water quality variables normalised (standardised) prior to analysis.

3 RESULTS AND DISCUSSION

3.1 Water quality

3.1.1 Physico-chemistry

All physico-chemical data collected are presented in Tables 3 to 8.

Water quality data were compared to the default ANZECC/ARMCANZ (2000) guidelines for the protection of aquatic systems in tropical Australia (Appendix 2). As already discussed in Section 2.3, the primary objective of the guidelines is to "maintain and enhance the 'ecological integrity' of freshwater and marine ecosystems, including biological diversity, relative abundance, and ecological processes" (ANZECC/ARMCANZ 2000). However, caution must be taken when applying trigger values to natural systems because the guidelines are generic and tend to be conservative. A recorded value outside the guidelines does not necessarily indicate anthropogenic disturbance, they are merely a 'trigger' for further investigation. ANZECC/ARMCANZ (2000) recommends developing system-specific guidelines in new areas for which there is adequate reference condition data to allow for specifics of water chemistry. The default TVs for physical and chemical stressors applicable to tropical northern Australia are provided in Appendix 2.

Dissolved oxygen

During the dry season, daytime dissolved oxygen levels ranged from 14% at KB50 to 124% at KB56 (Table 3 & Figure 3). In the wet season, DO levels ranged from 54% at JC91 to 107% at both KB50 and KBNW2 (Table 4 & Figure 3).

The majority of sites recorded DO values outside the ANZECC/ARMCANZ (2000) guidelines (Tables 3 & 4, and Figure 3). The 'high' DO values (>120%) recorded from KB56 and REF61 in the dry season are not likely to cause environmental concern, although they may reflect a site which experiences oxygen stress at night. It is likely that low water levels in the dry season, concomitant with higher light intensity led to increased algal and macrophyte growth, thereby resulting in higher DO levels. Supersaturated DO recorded from a number of sites in the dry season likely reflects excessive primary productivity. These sites likely go into oxygen stress at night, and may become anoxic as respiration by plants, algae and fauna depletes DO.

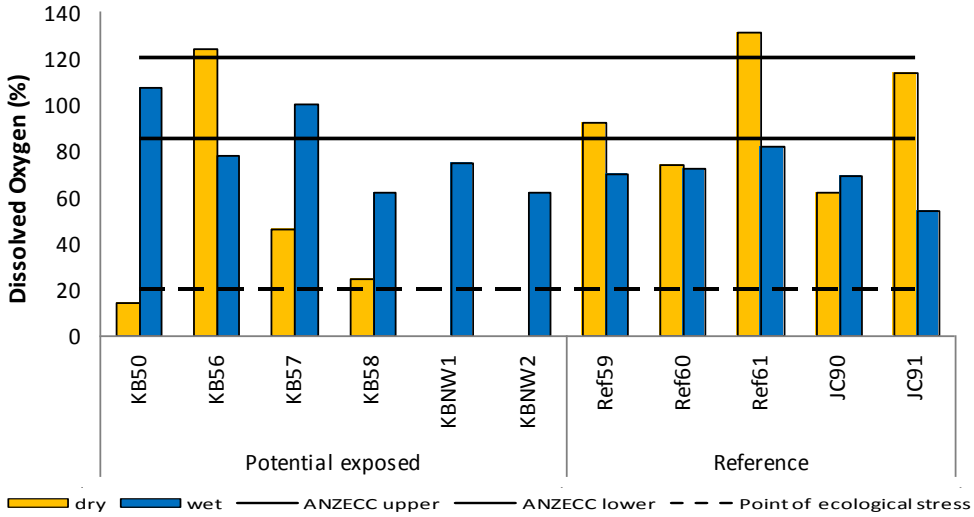


Figure 3. Dissolved oxygen (%) levels recorded from all WPIOP sites in the dry 2008 and wet 2009.

With the exception of Kens Bore 50, the 'low' DO levels recorded in the current study are not considered sufficiently low to have an ecological impact. DO concentrations less than ~20% typically represent environmental conditions of 'stress' to resident aquatic fauna, particularly fish with high metabolic demand for oxygen. During the dry season, Kens Bore 50 had receded to a small pool and recorded a DO saturation of 14%. At the time of sampling, fish were showing signs of stress and were air-breathing. Prolonged periods of low DO would have a detrimental effect to the ecosystem.

Table 3. *In situ* water quality data recorded from WPIOP sites in the dry season 2008. Shading indicates values outside ANZECC/ARMCANZ (2000) guidelines.

Site	Date	Time	pH	EC µS/cm	DO %	DO (mg/L)	Temp	Max depth (m)
KB50	14/12/08	700	8.02	608	14	1.2	20.3	0.50
KB56	13/12/08	1330	7.98	457	124	8.9	31.7	0.50
KB57	13/12/08	1100	7.73	543	46	3.9	24.1	0.40
KB58	13/12/08	830	8.10	1242	25	2.5	20.5	0.15
REF59	14/12/08	1030	8.79	336	92	7.4	25.1	3.90
REF60	14/12/08	1330	8.09	63	74	6.2	24.1	7.70
REF61	13/12/08	1600	9.62	144	131	10.5	27.4	3.50
JC90	16/12/08	0900	8.22	852	62	5.2	24.8	0.75
JC91	16/12/08	1130	9.14	908	114	91	25.7	1.95

Table 4. *In situ* water quality data recorded from WPIOP sites in the wet season 2009. Shading indicates values outside ANZECC/ARMCANZ (2000) guidelines.

Site	Date	Time	pH	EC µS/cm	DO %	DO (mg/L)	Temp	Max depth (m)
KB50	4/04/2009	1030	7.14	461	107	8.8	30.8	1.13
KB56	3/04/2009	1100	7.18	304	78	6.6	25.0	0.95
KB57	3/04/2009	1300	7.1	716	100	7.7	30.2	0.87
KB58	3/04/2009	800	7.28	305	62	5.3	22.4	1.05
KBNW1	5/04/2009	830	8.11	295	75	6.2	28.9	0.32
KBW2	5/04/2009	1130	7.10	475	62	5.4	26.8	1.02
REF59	4/04/2009	830	7.78	582	70	6.1	26.6	6.00
REF60	3/04/2009	1600	6.97	80	72	5.8	25.2	10.05
REF61	4/04/2009	1400	7.38	150	82	6.8	28.8	5.40
JC90	2/04/2009	0945	7.02	655	69	5.4	26.7	1.75
JC91	2/04/2009	1230	7.23	461	54	4.2	31.3	5.52

pH

Most river systems in Western Australia (including those in the Pilbara *e.g* Robe, Harding and lower Fortescue at Millstream) have a natural pH range circum-neutral. In the absence of baseline data, ANZECC/ARMCANZ (2000) guidelines recommend average pH should be between 6 and 8 in lowland rivers of tropical northern Australia. The pH values recorded during the current study were circum-neutral to basic and ranged from 6.97 (Reference site 60 in the wet 2009) to 9.62 (Reference site 61 in the dry 2008) (Tables 3 & 4, and Figure 4). Reference sites 59 and 61, and Jewel Cochrane sites 90 and 91 recorded pH values higher than the recommended ANZECC/ARMCANZ (2000) guidelines. The basic pH recorded from these sites is unlikely to cause impacts to biota and is similar to values previously recorded from the region (i.e. Johnson and Wright 2003, Streamtec 2004). The basic pH is considered a result of local geology and has been recorded from a number of other systems in the Pilbara Region, including Marillana Creek, Weeli Wolli Creek and Mindy Mindy Creek (WRM unpub. dat.).

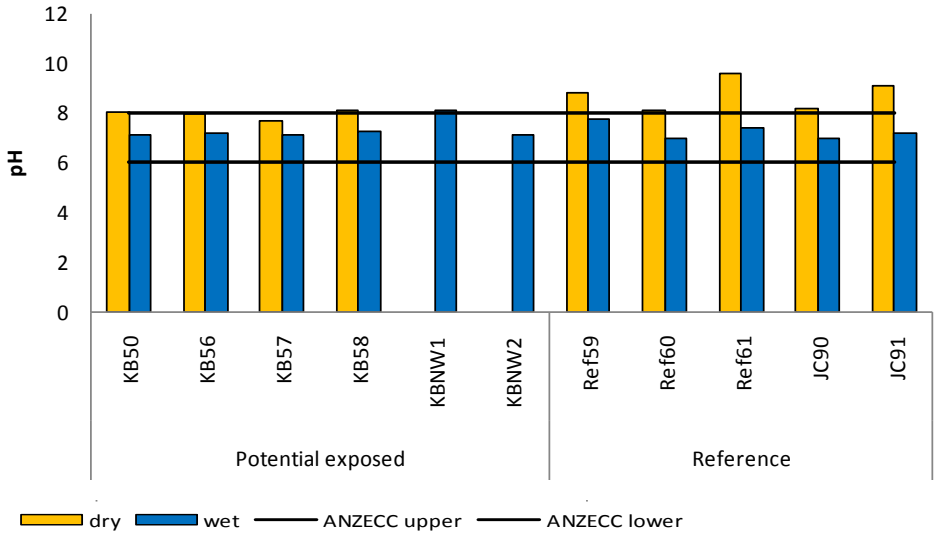


Figure 4. pH values recorded from WPIOP waterbodies in the dry 2008 and wet 2009.

Electrical conductivity

ANZECC/ARMCANZ (2000) guidelines were not used to make comparisons with electrical conductivity (Ec) in the current study, as the trigger value for freshwater rivers in tropical Australia is 250 µS/cm, and is not relevant to receding water holes in a semi-arid region.

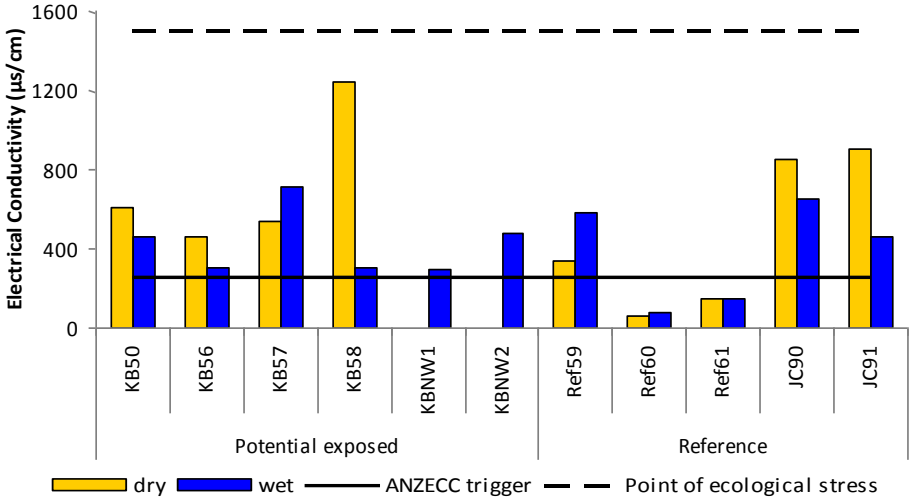


Figure 5. Electrical conductivity (µS/cm) recorded during WPIOP surveys in the dry 2008 and wet 2009.

Freshwater ecosystems are known to experience little ecological stress when Ec is less than 1500 µS/cm (Hart *et al.* 1991, Horrigan *et al.* 2005). Electrical conductivity did not exceed this value at any of the WPIOP sites sampled in either the wet or dry seasons (Figure 5).

All sites were considered fresh, as defined by ANZECC/ARMCANZ (2000)[3] (Tables 3 & 4). The generally higher Ec recorded in the dry season is likely a result of evapoconcentration effects.

Nutrients

Nutrient enrichment in aquatic systems can lead to increased algal growth and cyanobacterial blooms (ANZECC/ARMCANZ 2000), which may become more apparent as water levels

[3] Fresh defined as < 1500 µS/cm, Brackish = 1500 – 4500 µS/cm, Saline = 4500 – 50,000 µS/cm, Hypersaline > 50,000 µS/cm. A conversion factor of 0.68 was used to convert to conductivity µS/cm as recommended by ANZECC/ARMCANZ (2000).

recede, nutrients are evapo-concentrated, and water temperature increases. Such nuisance blooms can result in adverse impacts to the aquatic ecosystem through toxic effects, reductions in dissolved oxygen and changes in biodiversity (ANZECC/ARMCANZ 2000). Highly eutrophic waters tend to support high abundances of pollution-tolerant species, but few rare taxa, and overall, a less complex community structure.

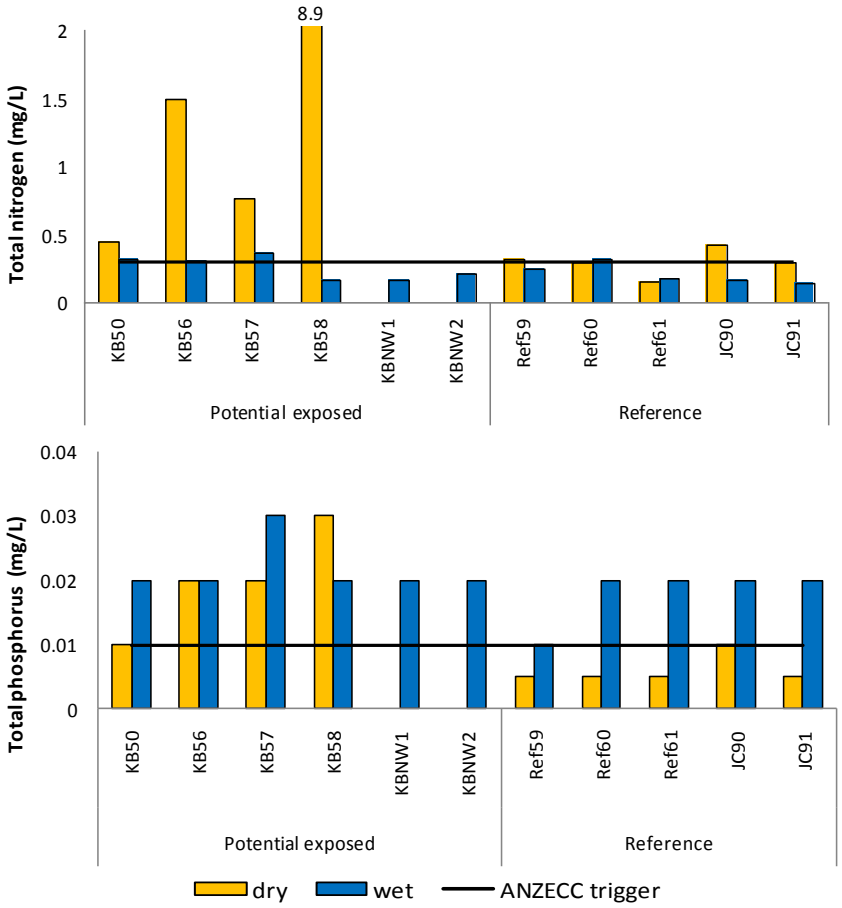


Figure 6. Total nitrogen (top) and total phosphorus levels (bottom) recorded from WPIOP waterbodies in the dry 2008 and wet 2009.

Total nitrogen exceeded ANZECC/ARMCANZ (2000) guidelines at six of the nine sites sampled in the dry, and four of the 11 sites sampled in the wet (Tables 5 & 6, and Figure 6). The total nitrogen concentration at the potential exposed site KB58 was exceptionally high; being more than 30 times the ANZECC/ARMCANZ (2000) trigger value. Total phosphorus was elevated at almost all sites in the wet season (Table 6 and Figure 6).

It should be noted that spot measurements of nutrients are not necessarily indicative of total nutrient loads. However, elevated nutrient levels at the end of the dry season are not unexpected given stock concentrating around water holes and general evapo-concentration of surface waters at this time of the year.

Ionic composition

The ionic composition of waters is determined by rain-borne salts (*i.e.* wind-blown dusts) and geology (*e.g.* weathering of soils) of the catchment (DeDeckker and Williams 1986). However, the composition over the warmer months, particularly in shallow reaches, will be altered by evapo-concentration and precipitation of less soluble salts, such as calcium carbonate and magnesium sulphate (Hart and McKelvie 1986). The ionic composition of inland waters in Australia is known to vary widely, but the proportions of calcium, magnesium and bicarbonate are often enriched compared to seawater (DeDeckker and Williams 1986).

The ionic dominance of waters at sampled sites was typically; Na>Ca>K>Mg: HCO3>Cl>SO4>CO3 (Tables 5 & 6). However, there was some spatial variability in both anion and cation dominance amongst sites and season (Tables 5 & 6).

Table 5. Nutrient and ionic composition data collected from WPIOP sites in the dry season of 2008. Shading indicates values outside ANZECC/ARMCANZ (2000) guidelines. All values are mg/L. Refer Table 3 for dates and times of water sample collection.

Site	Na	Mg	Ca	K	HCO3	CO3	Cl	SO4_S	Alkalinity	N_NH3	N_NO3	Total_N	P_SR	Total_P
KB50	55.1	23.4	22.4	10.3	165	<1	101	17.6	135	0.05	0.06	0.45	0.01	0.01
KB56	34.9	15.8	23.5	11.6	122	<1	79	12.6	100	0.39	<0.01	1.5	0.02	0.02
KB57	55.9	17.4	17.3	7.6	165	<1	87	13.9	135	0.04	0.06	0.76	0.01	0.02
KB58	92.6	42.9	59	32.9	305	<1	251	22.2	250	8.7	<0.01	8.9	0.01	0.03
REF59	37.1	10.2	13	5.7	122	<1	49	5	100	<0.01	<0.01	0.32	<0.01	<0.01
REF60	2.6	1.5	4.7	2	49	<1	5	<0.1	40	0.03	<0.01	0.3	<0.01	<0.01
REF61	12	3.5	7.1	5.6	31	18	18	2.1	55	<0.01	<0.01	0.15	<0.01	<0.01
JC90	78	37	34.3	8.7	287	<1	144	19.6	235	0.03	0.29	0.42	0.01	0.01
JC91	90.6	36.6	29.8	13.4	214	<1	193	22.7	175	<0.01	<0.01	0.3	<0.01	<0.01

Table 6. Nutrient and ionic composition data collected from WPIOP sites in the wet season of 2009. Shading indicates values outside ANZECC/ARMCANZ (2000) guidelines. All values are mg/L. Refer Table 4 for dates and times of water sample collection.

Site	Na	Mg	Ca	K	HCO3	CO3	Cl	SO4_S	Alkalinity	N_NH3	N_NO3	Total_N	P_SR	Total_P
KB50	42	16.3	15.7	5.8	128	<1	78	12.2	105	0.01	0.12	0.32	0.01	0.02
KB56	23.3	10	15.2	4.4	92	<1	46	8.1	75	0.01	0.05	0.31	0.01	0.02
KB57	47.6	21.2	23.4	6.8	165	<1	156	17	135	0.01	<0.01	0.36	0.01	0.03
KB58	25.7	10.7	14.7	4.7	122	<1	39	8.3	100	<0.01	<0.01	0.17	0.01	0.02
KBNW1	51.8	17.2	18.4	7.8	122	<1	135	17	100	<0.01	<0.01	0.25	0.01	0.01
KBNW2	37.2	20.8	30.7	5	177	<1	102	16.2	145	<0.01	<0.01	0.16	0.01	0.02
Ref59	38.5	16.3	25.4	7.2	116	<1	81	13.6	95	0.01	<0.01	0.21	0.01	0.02
Ref60	4.5	1.3	2.4	1.5	31	<1	11	1.9	25	0.03	0.05	0.32	<0.01	0.02
Ref61	15.7	3.4	5.7	3.7	55	<1	23	5	45	0.02	0.01	0.18	<0.01	0.02
JC90	45.6	18.7	18.2	6.4	201	<1	120	12.1	165	<0.01	0.04	0.16	0.01	0.02
JC91	44.2	15.5	15.7	5.8	140	<1	73	11.6	115	<0.01	0.01	0.14	0.01	0.02

Table 7. Metal concentration data collected from WPIOP aquatic survey sites in the dry season of 2008. Shading indicates values outside ANZECC/ARMCANZ (2000) guidelines. All values are mg/L.

	Al	As	B	Ba	Cd	Co	Cr	Cu	Fe	Mn	Mo	Ni	Pb	Se	U	V	Zn
KB50	<0.005	<0.001	0.18	0.17	<0.0001	<0.005	<0.001	<0.002	0.24	0.11	<0.001	<0.001	<0.0001	<0.001	<0.0001	<0.005	0.05
KB56	<0.005	<0.001	0.14	0.054	<0.0001	<0.005	<0.001	<0.002	0.18	0.15	<0.001	0.002	0.0002	<0.001	<0.0001	<0.005	0.011
KB57	<0.005	<0.001	0.23	0.048	<0.0001	<0.005	0.001	<0.002	0.053	0.11	<0.001	<0.001	0.0002	<0.001	<0.0001	<0.005	0.009
KB58	<0.005	0.002	0.33	0.17	<0.0001	<0.005	0.001	0.008	0.11	1.1	<0.001	0.002	0.0001	<0.001	0.0004	<0.005	<0.005
REF59	<0.005	<0.001	0.13	0.086	<0.0001	<0.005	<0.001	<0.002	<0.005	0.003	<0.001	<0.001	0.0002	<0.001	<0.0001	<0.005	0.042
REF60	<0.005	<0.001	<0.02	0.027	<0.0001	<0.005	<0.001	<0.002	<0.005	0.006	<0.001	<0.001	<0.0001	<0.001	<0.0001	<0.005	0.02
REF61	<0.005	<0.001	0.03	0.016	<0.0001	<0.005	<0.001	0.002	0.013	0.012	<0.001	<0.001	0.0001	<0.001	<0.0001	<0.005	0.022
JC90	<0.005	<0.001	0.24	0.03	<0.0001	<0.005	<0.001	<0.002	<0.005	0.015	<0.001	<0.001	<0.0001	<0.001	0.0005	<0.005	0.009
JC91	<0.005	<0.001	0.25	0.043	<0.0001	<0.005	<0.001	<0.002	<0.005	0.006	<0.001	<0.001	<0.0001	<0.001	0.0001	<0.005	0.045

Table 8. Metal concentration data collected from WPIOP aquatic survey sites in the wet season of 2009. Shading indicates values outside ANZECC/ARMCANZ (2000) guidelines. All values are mg/L.

	Al	As	B	Ba	Cd	Co	Cr	Cu	Fe	Mn	Mo	Ni	Pb	Se	U	V	Zn
KB50	<0.005	<0.001	0.17	0.033	<0.0001	<0.0001	<0.0005	0.002	0.087	0.016	<0.001	<0.001	0.0001	<0.001	<0.0001	<0.005	0.006
KB56	<0.005	<0.001	0.11	0.034	<0.0001	<0.0001	<0.0005	0.0024	0.059	0.021	<0.001	<0.001	<0.0001	<0.001	<0.0001	<0.005	0.007
KB57	<0.005	<0.001	0.22	0.05	<0.0001	<0.0001	<0.0005	0.0009	0.008	0.015	<0.001	<0.001	<0.0001	<0.001	<0.0001	<0.005	0.005
KB58	<0.005	<0.001	0.12	0.028	<0.0001	0.0001	<0.0005	0.001	0.054	0.027	<0.001	<0.001	<0.0001	<0.001	<0.0001	<0.005	0.008
KBNW1	<0.005	<0.001	0.2	0.043	<0.0001	<0.0001	<0.0005	0.0012	0.011	0.006	<0.001	<0.001	0.0002	<0.001	<0.0001	<0.005	0.01
KBNW2	<0.005	<0.001	0.14	0.039	<0.0001	<0.0001	<0.0005	0.0006	0.022	0.012	<0.001	<0.001	<0.0001	<0.001	0.0004	<0.005	0.007
Ref59	<0.005	<0.001	0.14	0.044	<0.0001	<0.0001	<0.0005	0.0018	0.022	0.005	<0.001	<0.001	0.0001	<0.001	<0.0001	<0.005	0.004
Ref60	<0.005	<0.001	0.02	0.008	<0.0001	<0.0001	<0.0005	0.0002	0.058	0.044	<0.001	<0.001	<0.0001	<0.001	<0.0001	<0.005	0.004
Ref61	0.006	<0.001	0.07	0.024	<0.0001	0.0001	<0.0005	0.002	0.26	0.1	<0.001	<0.001	0.0001	<0.001	<0.0001	<0.005	0.007
JC90	<0.005	<0.001	0.15	0.021	<0.0001	<0.0001	<0.0005	0.0014	0.01	<0.001	<0.001	<0.001	<0.0001	<0.001	<0.0001	<0.005	0.006
JC91	<0.005	<0.001	0.14	0.02	<0.0001	<0.0001	<0.0005	0.002	0.023	0.004	<0.001	<0.001	0.0001	<0.001	<0.0001	<0.005	0.005

Metals

Metal pollution is known to adversely impact aquatic biota; especially populations of metal-sensitive groups such as crustaceans (e.g. Hynes 1970). Metal levels were generally low (Tables 7 and 8); however, levels of boron, copper and zinc exceeded ANZECC/ARMCANZ (2000) guidelines for the protection of 99% of species at some sites (Appendix 2).

Copper exceeded ANZECC/ARMCANZ (2000) guidelines at a number of sites, including the potential exposed sites KB50 (0.002[4] in the wet), KB56 (0.0024 in the wet), and KB58 (0.008 mg/L in the dry), and the reference sites REF59 (0.0018 in the wet), REF61 (0.002 in the dry), and JC91 (0.002 in the dry). Copper concentrations were relatively high at KB58 in the dry season (Figure 7).

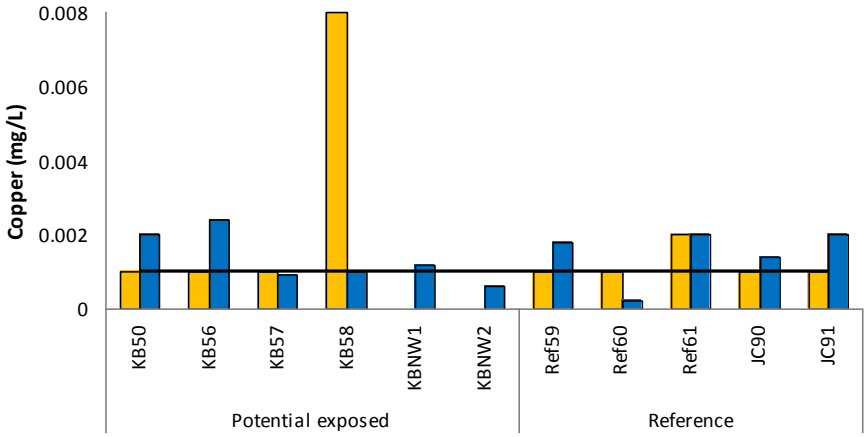


Figure 7. Copper concentrations at WPIOP sites in the dry 2008 and wet 2009.

Copper can be highly toxic in aquatic environments and can adversely affect algae, invertebrates, fish, amphibians and water birds (Owen 1981). Elevated copper levels have been shown to lead to reductions in overall macroinvertebrate richness, particularly in sensitive 'EPT' (**E**phemeroptera, **P**lecoptera and **T**richoptera) taxa (Malmqvist and Hoffsten 1999). However, copper concentrations of the magnitude recorded during the current study are unlikely to adversely impact aquatic biota. They also represent background levels for this system and these data may be used to develop a system-specific TV for Cu.

Zinc exceeded ANZECC/ ARMCANZ (2000) guidelines at all sites[5] in both seasons; except KB58 in the dry season which recorded zinc at the same level as the TV (see Tables 7 & 8, and Figure 8). At the potential exposed site KB50 and reference sites REF59 and JC91 in particular, zinc levels were highest, exceeding the ANZECC/ ARMCANZ (2000) guidelines by over 16 times. At these concentrations, zinc can become toxic to aquatic organisms, particularly crustaceans and molluscs.

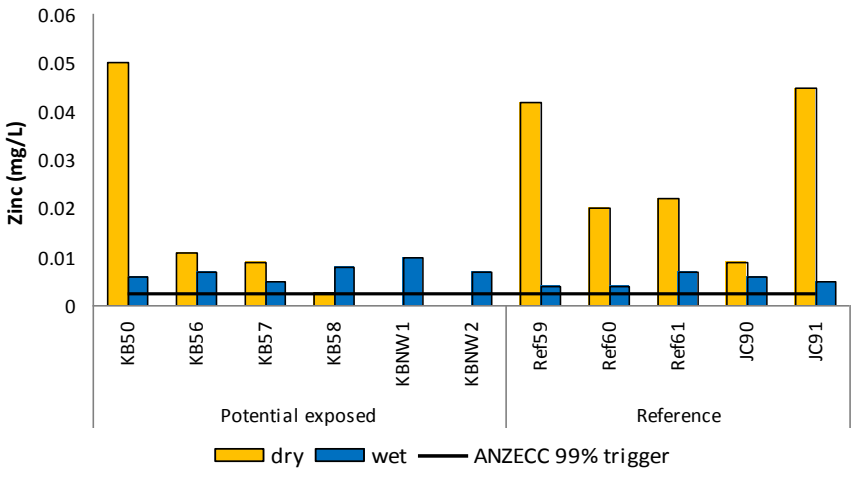


Figure 8. Zinc concentrations at WPIOP sites in the dry 2008 and wet 2009.

[4] 99% trigger value for copper = 0.001 mg/L
[5] 99% trigger value for zinc = 0.0024 mg/L

The high levels of some heavy metals were considered due to local geology. A number of heavy metals occur naturally in sediment, including mercury, cadmium, copper and zinc, and the concentration of such metals can build up over time through natural processes. Elevated levels of zinc and copper have previously been recorded from waterbodies in the Pilbara Region (Streamtec 2004, WRM unpub. data).

Even though elevated, it is unknown what proportion of the measured dissolved metals was labile (bio-available) or unavailable through complexing with dissolved organic carbon (*e.g.* tannin). The bioavailability of trace metals is affected by a number of factors including, water hardness (Stephenson and Mackie 1989), alkalinity, salinity (Jackson *et al.* 2000), pH (Jackson *et al.* 2000) as well as what chemical form the metal is in (Sander *et al.* 2007). Zinc is an essential micronutrient, whereas cadmium is extremely toxic, but when they occur in the same environment there is potential for the two metals to compete for the same biological binding sites. In a study of the complexation of Cd and Zn in alpine lakes in New Zealand, Sander *et al.* (2007) found that despite cadmium being recorded in much lower total concentrations than copper and zinc, it bears the highest toxicity for aquatic organisms.

ANZECC/ARMCANZ (2000) recommend the use of techniques such as DGTs (Diffuse Gradients in Thin Films) as a speciation measurement to provide a better estimate of the bio-available metal concentration if the dissolved metal concentrations exceed the guideline trigger values. Though aquatic biota may be adapted to elevated background levels of some heavy metals, any further increase due to mine activities may exceed tolerance thresholds and result in loss of biodiversity. Tolerance thresholds for the majority of Western Australia's aquatic biota are unknown. These data provide an indication of background levels and establish that some metals are naturally elevated, which is important to document prior to mine development.

Diffuse Gradients in Thin Films (DGTs).

The DGT technique was first developed in 1994 as a time averaged, *in situ* speciation measurement of heavy metals in waters. Since its introduction it has been validated in the field for the determination of metals in fresh and seawater, and more recently in estuarine waters. The DGT technique is based on a simple device, which accumulates metal ions in a well-defined manner from solution. Soluble species diffuse through a diffusive layer of known thickness in which a concentration gradient is maintained. Behind the diffusive layer is a binding layer in which reactive metal species are bound. The mass of accumulated metal is measured following retrieval and is used to calculate the average concentration of DGT labile metal species in the bulk solution over the deployment time. As the device does not accumulate the major ions that cause interference with the measurement, the measurement does not suffer the degree of interference associated with the direct analysis of waters.

3.1.2 Patterns in water quality data

MDS ordination indicated a significant difference in overall water quality between seasons (ANOSIM; Global R = 0.334, significance level of sample statistic, p = 0.0002; Figure 9), but not treatment type (ANOSIM; Global R = 0.106, significance level of sample statistic, p = 0.025[6]; Figure 9). Generally, the water quality of reference sites was more variable than that of potential exposed sites (Figure 9). This likely reflects the greater range in types of habitats and geographical location of sites sampled for references sites (seasonal river pools, deep permanent pools below falls) compared with potentially exposed sites.

[6] Although the p-value was significant the Global R-value of 0.106 indicates that the groups were barely separable.

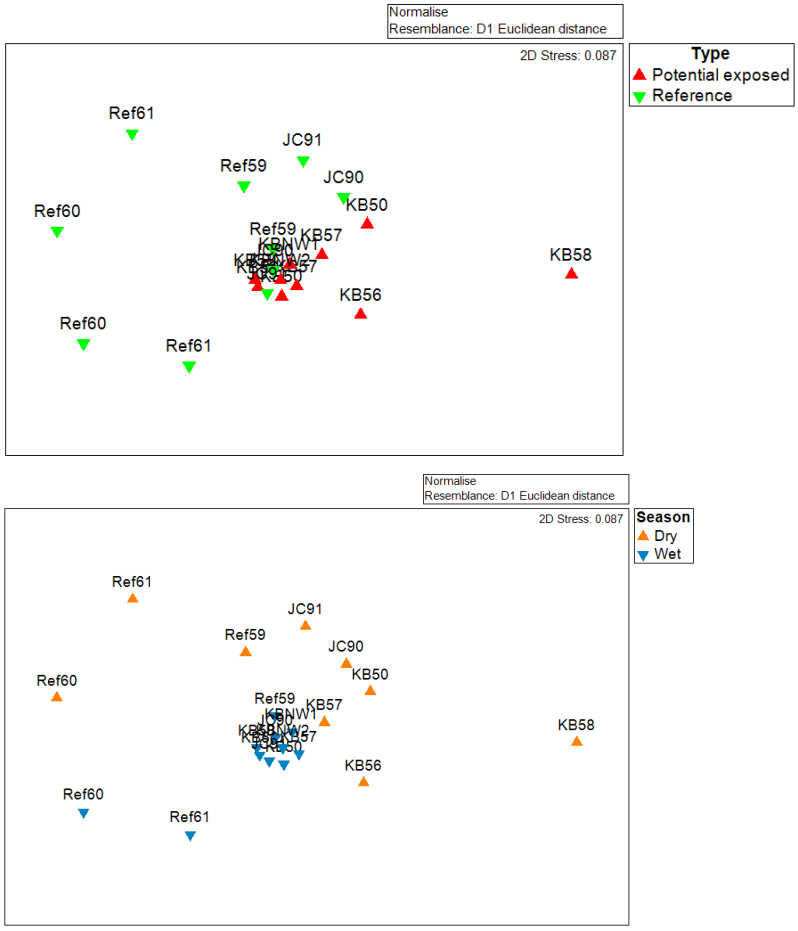


Figure 9. MDS plot of environmental (water quality) data (normalised & some variables log transformed) based on a Euclidean distance measure. Samples are coloured by type (top) and season (bottom). Stress was low (0.09).

3.2 Microinvertebrates

3.2.1 Taxonomic composition and species richness

A total of 135 taxa of microinvertebrates were recorded from WPIOP waterbodies across the wet and dry seasons of 2008/2009 (Appendix 3). The microinvertebrate fauna comprised 27 types of Protista (two Ciliophora, and 25 Rhizopoda), 81 taxa of Rotifera (seven Bdelloidea, and 74 Monogononta), 11 types of Cladocera (water fleas), 11 species of Copepoda and five taxa of Ostracoda (seed shrimp).

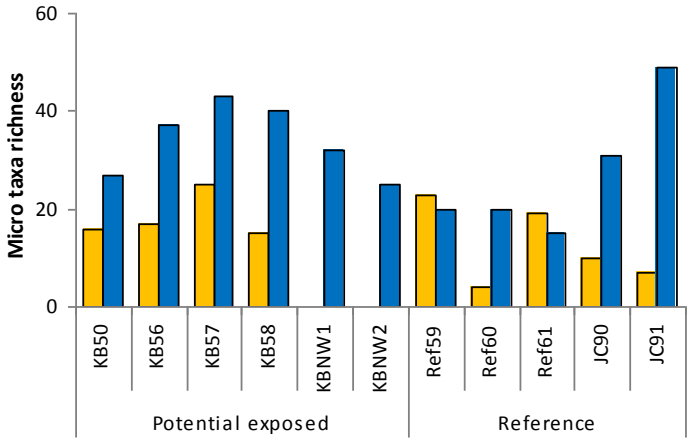


Figure 10. Microinvertebrate taxa richness from each site.

Of the 135 taxa, only two were commonly encountered, occurring in over 75% of samples (i.e. ≥ 15 samples). These were the

Protista *Arcella discoides* and Copepod cyclopoid nauplii (Appendix 3). In contrast, 46 taxa were uncommon and were only recorded from one sample.

Taxa richness varied between site and season (Figures 10 and 11). The greatest number of microinvertebrate taxa was collected from the reference site JC91 during the wet season (49 taxa) and the least collected from REF60 in the dry (4 taxa; see Figure 10). There was a significant difference in the number of microinvertebrate taxa recorded between seasons, but not between types (Table 9 and Figure 11). There was no interaction between season and type (Table 9 and Figure 11).

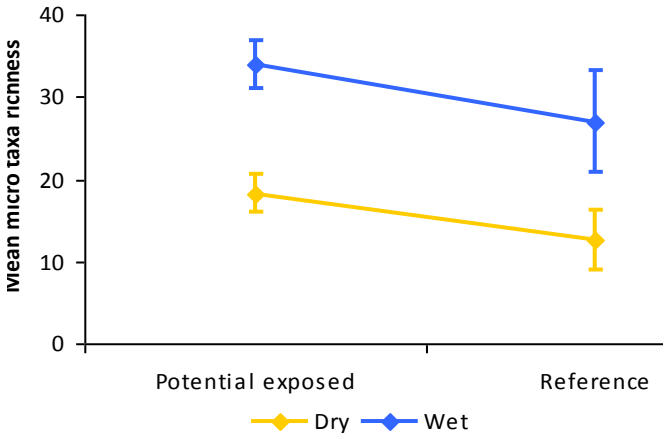


Figure 11. Mean microinvertebrate taxa richness (±se) for each treatment type in each season.

Table 9. Two-way ANOVA results for microinvertebrate taxa richness by season and type.

Source	df	Mean Square	F	p-value
Season	1	1113.09	13.43	0.002
Type	1	195.95	2.36	0.144
Season *type	1	2.23	0.03	0.872
Error	16	82.87		
Total	19			

3.2.2 Conservation significance of microinvertebrates

The majority of microinvertebrate taxa recorded were common, ubiquitous species. Of the 135 microinvertebrate taxa recorded from WPIOP in the dry season of 2008 and wet 2009, 51% were cosmopolitan species, occurring widely throughout the world, 4% had a pan-tropical distribution, and 1% were Australasian species. Over 40% of taxa were indeterminate due to insufficient information/taxonomy. Of interest however, was the collection of species only known from the Australian Continent (4% of taxa). Australian endemic species included the protist *Difflugia australis*, the rotifers *Lecane noobijupi* and *Lecane batillifer*, and the Cladocera *Alona rigidicaudis* (Plate 4) and *Moina* cf. *micrura*. The former species, *D. australis*, has previously been recorded from Kakadu in the Northern Territory, north-west W.A., and during sampling undertaken by the authors was collected from a number of systems in the Pilbara Region, including the Fortescue River (Roy Hill), Marillana Creek, Coondiner Creek, and a tributary of Mindy Mindy Creek (WRM unpub. data). During the current study, this species was collected from the potential exposed sites KB56, KB57, KB58 and KBNW1, as well as the reference site REF60 (Appendix 3). The rotifer *L. noobijupi* has been recently recorded

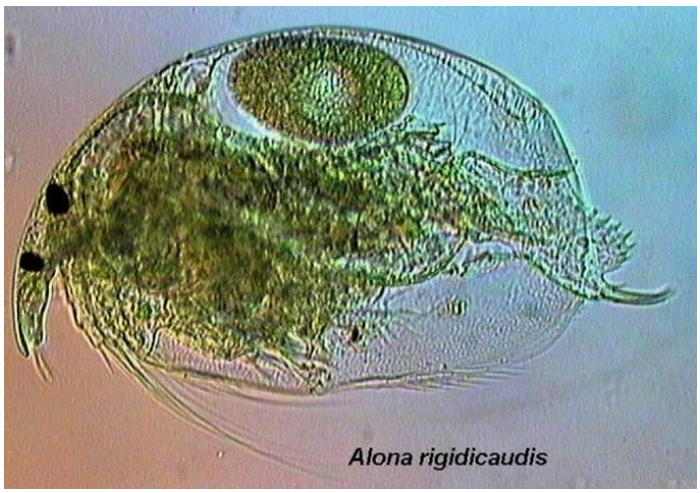


Plate 4. *Alona rigidicaudis* (photo by Russ Shiel).

from Weeli Wolli Creek and Mindy Mindy Creek (WRM unpub. data), and has previously been recorded from a number of locations throughout the south-west of the State (Segers and Shiel 2003).

Also of interest within the microinvertebrate fauna were two species which were classified as new records for either Australia or Western Australia. The collection of the protist *Difflugia capreolata* from the potential exposed site KBNW2 was a new record for Australia. In addition, the rotifer *Cephalodella gigantea* collected from the reference site JC91 constituted a new record for Western Australia. It should be noted that collecting efforts for microinvertebrate fauna are relatively recent, and likely to produce a species list, most of which on probability are likely to be known taxa, but a small proportion of which are likely to be unknown.

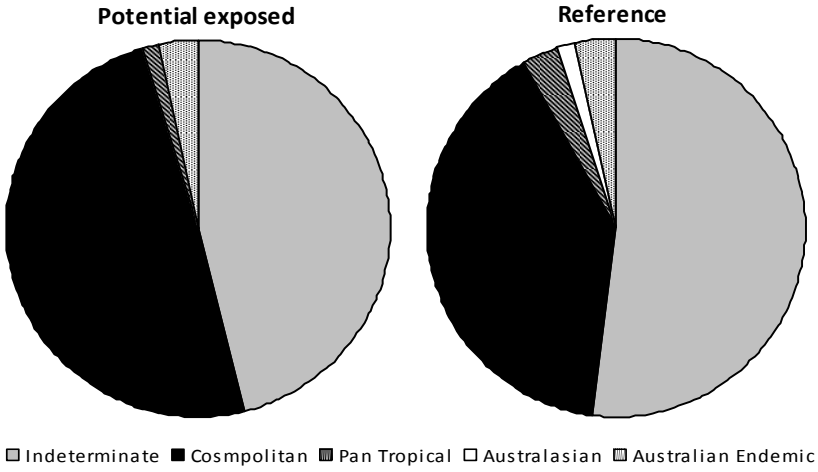


Figure 12. Pie-charts of the proportion of species from each conservation category for potential exposed sites (left), and reference sites (right).

Endemicity of micro-fauna was similar between potential exposed sites and reference sites (Figure 12). Cosmopolitan and indeterminate taxa made up the greatest proportions from each type (Figure 12). Australian endemic species were recorded in low proportions from both potential exposed and reference sites (Figure 12). Australasian species were only recorded from reference sites.

3.2.3 Patterns in microinvertebrate assemblage structure

The Bray-Curtis similarity matrices for dry and wet season samples are provided in Tables 10 and 11, respectively. Microinvertebrate assemblages were significantly different between type (ANOSIM; Global R = 0.515[7], significance level of sample statistic, p = 0.002; Figure 13). The wet season microinvertebrate assemblages of the two Jewel Cochrane sites (reference sites) were more similar to the wet season Kens Bore sites (potential exposed). There was however, no significant difference in microinvertebrate assemblages between season (ANOSIM; Global R = 0.314, significance level of sample statistic, p = 0.009; Figure 13).

Given that microinvertebrate assemblages are different between potential exposed and reference sites prior to the commencement of mining, relative changes in Bray-Curtis similarities can be used in future monitoring to assess any effect of dewatering and possible discharge operations.

[7] Global R = degree of separation between groups, where R-statistic >0.75 = groups well separated, R-statistic >0.5 = groups overlapping but clearly different, and R-statistic >0.25 = groups barely separable.

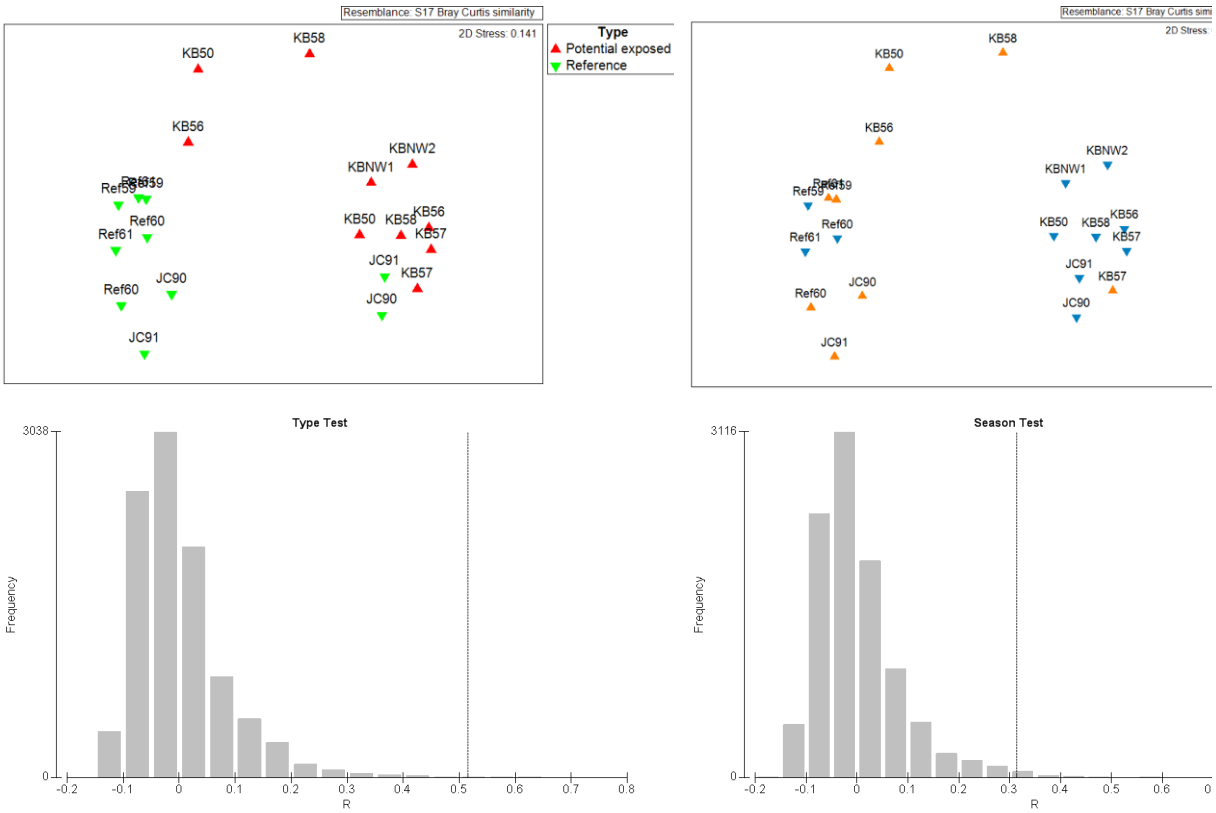


Figure 13. MDS plot of microinvertebrate data using on log_{10} abundance categories and based on Bray-Curtis Similarity. Samples are coloured by type (top left) and season (top right). Stress was 0.14. ANOSIM plots are provided for the type test (bottom left) and season test (bottom right), showing that the R-value falls within the range of expected R-values for season (i.e. not significantly different), but not type.

Table 10. Bray-Curtis similarity matrix for dry season microinvertebrate samples.

	KB50	KB56	KB57	KB58	Ref59	Ref60	Ref61	JC90
KB50								
KB56	49.12							
KB57	15.79	17.39						
KB58	34.48	27.45	22.86					
Ref59	40.54	29.85	11.63	20.59				
Ref60	18.60	27.78	10.91	16.22	41.51			
Ref61	33.33	30.51	12.82	10.00	63.16	40.00		
JC90	15.69	31.82	22.22	22.22	29.51	53.33	26.42	
JC91	16.67	29.27	13.33	14.29	24.14	51.85	24.00	45.71

Table 11. Bray-Curtis similarity matrix for wet season microinvertebrate samples.

	KB50	KB56	KB57	KB58	Ref59	Ref60	Ref61	JC90	JC91	KBNW 1
KB50										
KB56	38.53									
KB57	38.94	45.21								
KB58	45.71	50.72	46.48							
Ref59	16.44	11.32	12.73	13.73						
Ref60	21.05	20.18	21.24	20.95	49.32					
Ref61	17.39	15.69	13.21	14.29	42.42	49.28				
JC90	29.41	35.56	31.65	39.69	16.16	15.69	16.84			
JC91	34.15	42.31	40.00	43.42	15.00	16.26	18.97	51.01		
KBNW 1	27.96	36.51	32.31	42.62	17.78	19.35	16.28	38.66	41.43	
KBNW 2	35.90	30.63	38.26	39.25	18.67	17.95	11.27	25.00	25.60	40.00

3.3 Hyporheic fauna

3.3.1 Taxonomic composition and species richness

Three of the reference sites (REF59, REF60 and REF61) could not be sampled for hyporheic fauna due to the lack of appropriate substrate (gravel) and dominance by rocky outcrops.

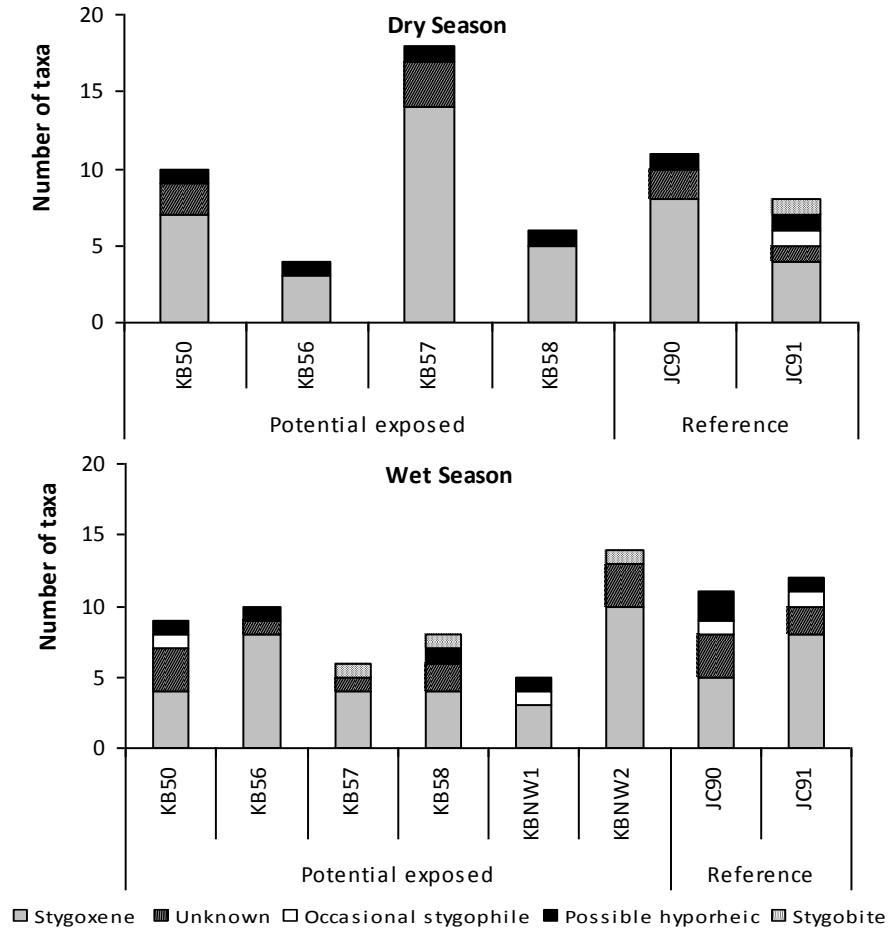


Figure 14. Number of taxa from each hyporheic classification category. Data collected in the dry season 2008 (top), and wet season 2009 (bottom).

Sampling of hyporheic habitats from seven sites in the WPIOP study area in the dry season of 2008, and nine sites in the wet season 2009 revealed the presence of hyporheic fauna (Appendix 4). Of the 45 taxa collected in the hyporheic samples, the vast majority were classified as stygoxene (69%), i.e. they do not have specialised adaptations for groundwater habitats and were likely surface forms present in samples. However, 2% of the taxa were classified as occasional

hyporheos stygophiles, 2% were considered true stygobites, and 5% were considered possible hyporheic fauna. None were considered to be permanent hyporheos stygophiles and 22% were unknown due to insufficient taxonomy and/or information (Appendix 4). Therefore, 9% of taxa collected from hyporheic samples were likely hyporheos taxa and adapted to life in the hyporheic zone. Classifications followed those by Boulton (2001), however this type of analysis should be treated with some caution as results are likely affected by available information on life history, taxonomic resolution, and interpretation of classification categories.

Species considered to be hyporheos were the stygobitic amphipod ?Paramelitidae spp., occasional stygophile copepod *Microcyclops varicans*, and possible hyporheic taxa Nematoda spp., and Oligochaeta spp. Hyporheic fauna were collected from all sites in both the wet and dry seasons (Figure 14).

The results from this initial survey are similar to those reported previously in the Pilbara (Halse *et al.* 2002, WRM unpub. data), in that <20% of taxa collected in hyporheic habitats were entirely dependent on groundwater for their persistence as a species. Halse *et al.* (2002) suggested that it is not surprising that the hyporheos is dominated by species with some affinity for surface water, because the hyporheos is an "ecotone between productive, species-rich surface water systems and nutrient-poor groundwater systems with lower number of species per sampling unit".
Hyporheic fauna distribution and composition tend to be highly variable both temporally and spatially. Changes in composition can occur vertically within the riverbed and laterally towards the floodplain (Pennak and Ward 1986). Patchy distributions of the hyporheos can also be caused by event-induced and
site-specific discontinuities which are characteristic of the hyporheic interstices (Danielopol 1991, Marmonier *et al.* 1992). The two most important factors which control hyporheos distributions
and composition include sediment porosity, which influences interstitial volume and surface area (useable pore space, viz. habitat area), and hydraulic conductivity, which influences interstitial flow and consequently variables such as oxygen content, temperature and organic matter storage (Findlay 1995, Brunke and Gonser 1997).

3.3.2 Hyporheos taxa

Stygobitic amphipods were recorded from a number of sites in the WPIOP study area, including the potential exposed sites KB57, KB58, and KBNW2, and reference site JC91 (Appendix 4). Although these specimens are still awaiting formal identification, they were known to be restricted to groundwater and hyporheic environments because they exhibited a number of characteristics unique to stygofauna, including a lack of pigmentation, eyes reduced or absent, and long appendages. WRM have previously recorded stygobitic amphipods from both Weeli Wolli Creek and Marillana Creek. These specimens were known to be of the genus *Chydaekata* (Family: Paramelitidae), but were unable to be identified further using morphological taxonomy. The stygobitic amphipods from WPIOP may be formally identified through genetic analysis using DNA, although this was beyond the scope of the current study.

The copepod *Microcyclops varicans* was classified as an occasional hyporheic stygophile as it is known from groundwater environments, but is also widespread in surface waters. This species has been recorded from groundwaters in the Gascoyne (Box Well on House Station and Two-Mile Bore on Killara Station; Karanovic 2004), and Pilbara (bores at Newman and Mulga Downs; Karanovic 2006). During the current study this species was recorded from the hyporheic zone of potential exposed sites KB50 and KBNW1, as well as reference sites JC90 and

JC91. Previously, this species has been recorded by the authors from Weeli Wolli Creek (WRM unpub. data).

The Oligochaetes (segmented freshwater worms) collected during the current study were classified as possible hyporheic fauna. This is because a number of species known to occur in hyporheic and groundwater environments have previously been collected by the authors from other systems in the Pilbara Region including Weeli Wolli Creek, Marillana Creek, and Coondiner Creek (WRM unpub. data). It is unknown whether the oligochaete specimens collected from the WPIOP study area can be classified as hyporheos species because they have not been formally identified. Oligochaetes require identification by taxonomic experts.

Similarly, the Nematodes (freshwater round worms) collected from WPIOP hyporheic samples were considered as possible hyporheic fauna. Identification of Western Australian nematodes to species-level is constrained by limitations in taxonomy. The EPA (2007) recognised the difficulty in identifying Western Australian nematodes in their draft guidance statement for sampling stygofauna in Western Australia. While identification to species-level is generally required to understand an individual's ecological dependence on groundwater or the hyporheic zone, a number of Western Australian studies have reported the presence of nematoda considered to be groundwater species (Eberhard *et al.* 2005, Boulton *et al.* 2008, Eberhard *et al.* 2009). Therefore, nematodes collected from the hyporheic zone in the current study were classified as possible hyporheic fauna.

3.4 Macroinvertebrates

3.4.1 Taxonomic composition and species richness

A total of 128 taxa of macroinvertebrates were recorded from the twenty sites sampled in the dry season of 2008 and wet of 2009 (Appendix 5). Within the macroinvertebrate fauna there were Nematoda (round worms), Hydrazoa (freshwater hydra), Oligochaeta (aquatic segmented worms), four species of Gastropoda (freshwater snails), Arachnida (water mites), five types of Ephemeroptera (mayflies), nine Odonata (dragonflies & damselflies), 14 Hemiptera (true bugs), 44 taxa of Coleoptera (aquatic beetles), 41 Diptera (aquatic fly larvae), five taxa of Trichoptera (caddis flies) and Nymphulinae (moth larvae).

The taxonomic listing also includes records of larval and pupal stages for groups such as Diptera and Coleoptera. Current taxonomy is not sufficiently developed to allow identification of larval and pupal stages of all members of these groups to species level. In many instances, it is likely that these stages are the same species as the larval/adult stages recorded from the same location. However, because this could not be definitively determined, they were treated as separate taxa. In any case, different life stages often have different ecological roles in the ecosystem, providing good functional reasons to treat them as separate taxa.

The composition of macroinvertebrate taxa was typical of freshwater systems throughout the world (Hynes 1970), and was dominated by Insecta (84% of taxa). Of the insects, the majority were Coleoptera (37% of Insecta), closely followed by Diptera (34% of Insecta). Molluscs only comprised 3% of the total fauna.

Of the 128 taxa, six were commonly encountered and occurred in over 75% of samples (i.e. ≥ 15 samples). These were Hydracarina spp., the mayfly *Tasmanocoenis arcuata*, ceratopogonid spp. larvae, and the chironomids *Procladius* sp., *Tanytarsus* sp., and chironomid pupae. In contrast, a total of 35 taxa were uncommon and only recorded once (from one sample; Appendix 5).

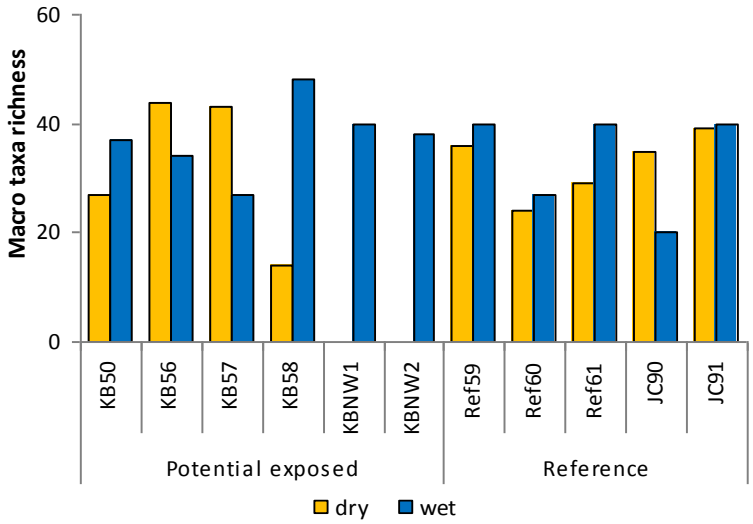


Figure 15. Macroinvertebrate taxa richness from each site.

Macroinvertebrate taxa richness varied between sites and seasons (Figure 15). The lowest number of macroinvertebrate taxa was collected from KB58 in the dry season (14 taxa; Figure 15). Interestingly, this site also recorded the greatest number of macroinvertebrate taxa, with 48 taxa being collected during the wet season (Figure 15 & Appendix 5).

Although the mean number of macroinvertebrate taxa collected was slightly higher in the wet season, there was no significant difference in taxa richness between seasons (Table 12 & Figure 16). Nor was there any significant difference in the number of taxa collected between site type (i.e. potential exposed vs reference sites; Table 12 and Figure 16). There was no interaction between season and type (Table 12).

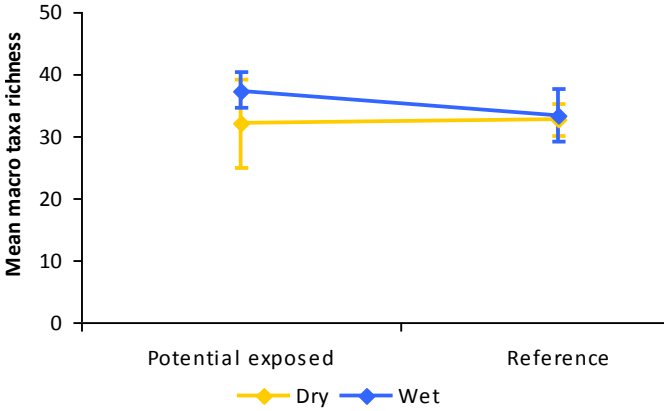


Figure 16. Mean macroinvertebrate taxa richness (±se) for each treatment type in each season.

Table 12. Two-way ANOVA results for macroinvertebrate taxa richness by season and type.

Source	df	Mean Square	F	p-value
Season	1	46.06	0.55	0.47
Type	1	13.61	0.16	0.69
Season *type	1	25.17	0.30	0.59
Error	16	84.36		
Total	19			

3.4.2 Conservation significance of macroinvertebrates

The majority of macroinvertebrate taxa recorded were common, ubiquitous species. Of the 128 macroinvertebrate taxa recorded from the WPIOP study area, 12% were cosmopolitan species, occurring widely throughout the world, 26% were Australasian, 4% had a northern Australian distribution, and 4% were endemic to Australia. Over 68% of taxa were indeterminate due to insufficient information/taxonomy. Of interest however, was the collection of species known only from the Pilbara region of Western Australia. Pilbara endemic species included the Odonate *Nannophlebia injibandi* and the haliplid beetle *Haliplus pilbaraensis*. Both species were collected from the potential exposed site KB58.

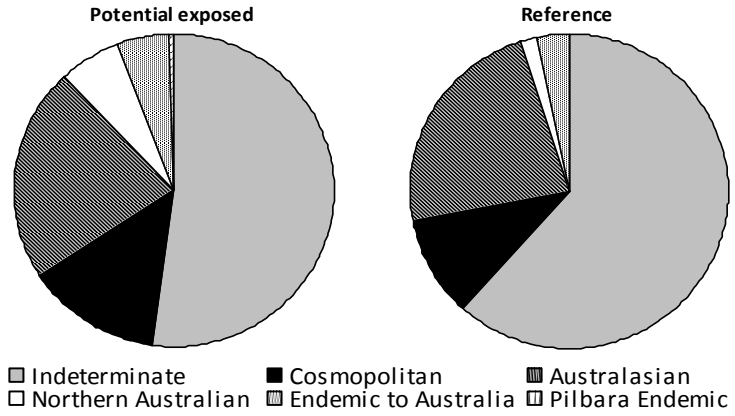


Figure 17. Pie-charts showing the proportion of macroinvertebrate species from each conservation category within each river system sampled.

The haliplid beetle *Haliplus pilbaraensis* has only recently been formally described (Dr Chris Watts, SAM, unpub. data). The holotype for this species was collected from Kalgan Pool on Kalgan Creek. It has been recorded widely across the Pilbara Region by CALM (now DEC), Chris Watts, the Western Australian Museum, and the authors during previous studies (WRM unpub. data).

Indeterminate taxa made up the greatest proportion of taxa from each site type (potential exposed vs reference; Figure 17). Northern Australian species and Australian endemics were recorded from both potential exposed and reference sites. However, species endemic to the Pilbara region were only recorded from potential exposed sites.

3.4.3 Functional feeding groups

It is generally considered that the functional complexity and 'health' of an aquatic ecosystem is reflected by the diversity of functional feeding groups[8] present (groups that reflect the obligate feeding mode of each species) (Cummins *et al.* 1995). As a result, aquatic macroinvertebrates are often classified into functional feeding groups, which reflect the mode of feeding by individual species. These groups include shredders, predators, filterers, grazers and collectors. The functional composition (i.e. the proportions of these groups) may be used to infer ecological health, whereby an ecologically healthy system has a mix of the different groups present. Covich *et al.* (1999) suggested that if each functional group is present in a system, ecological processes and energy flow are maintained.

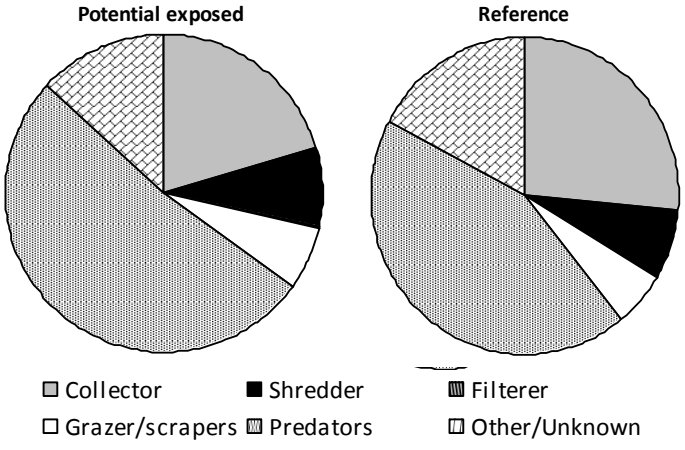


Figure 18. Pie-charts showing the proportion of macro-invertebrate species from each functional feeding group for potential exposed sites (left) and reference sites (right).

The functional organisation of macroinvertebrate communities was similar between potential exposed and reference sites at WPIOP (Figure 18). Predators were the dominant feeding group, followed by collectors (Figure 18). No filter-feeders were collected from potential exposed or reference sites (Figure 18). This is not entirely unexpected as the creek systems were not flowing during either the dry or wet seasons at WPIOP. The river systems were instead a series of pools. Filterers generally inhabit

[8] Functional feeding groups: 'shredders' feed on coarse particulate matter (CPOM >1mm); 'collector's feed on fine particulate matter (FPOM < 1mm); 'filterers' filter suspended particles from the water column and are often viewed as a subset of collectors; 'grazers' are those animals that graze or scrape algae and diatoms attached to the substrate; 'predators' capture live prey.

faster flowing waters where they can easily filter food particles and oxygen from the water as it flows past.

3.4.4 Patterns in macroinvertebrate assemblage structure

The Bray-Curtis similarity matrices for dry and wet season samples are provided in Tables 13 and 14, respectively.

Table 13. Bray-Curtis similarity matrix for dry season macroinvertebrate samples.

	KB50	KB56	KB57	KB58	Ref59	Ref60	Ref61	JC90
KB50								
KB56	47.37							
KB57	44.87	66.30						
KB58	31.46	39.32	33.06					
Ref59	30.88	36.59	45.24	35.64				
Ref60	36.89	35.11	35.56	38.24	40.00			
Ref61	38.40	47.06	49.68	37.78	64.23	40.38		
JC90	50.70	52.94	54.02	41.12	54.55	44.63	54.55	
JC91	46.36	43.58	38.25	34.48	45.40	41.54	51.32	68.64

Although there seemed to be some broad groupings evident within the MDS ordination, with potential exposed sites on the left and reference sites on the right of the ordination plot, the within type variation was high, i.e. macroinvertebrate assemblages were highly variable among potential exposed sites, and highly variable amongst reference sites (Figure 19). Therefore, there was no significant difference between macroinvertebrate assemblages of potential exposed sites vs reference sites (ANOSIM; Global R = 0.314, significance level of sample statistic, p = 0.0003[9]; Figure 19). This is a good result in terms of long-term monitoring as it will better enable testing for effects of mining. Because the macroinvertebrate assemblages of all sites are currently similar / indistinguishable, any future separation of exposed sites from reference sites would signal some effect.

There was a significant difference in macroinvertebrate assemblages between season (ANOSIM; Global R = 0.436, significance level of sample statistic, p = 0.0001; Figure 19.

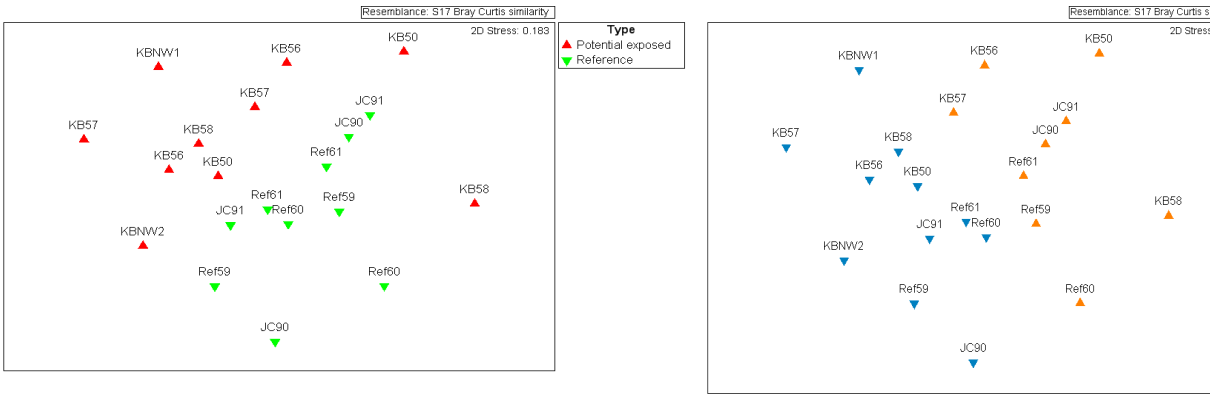


Figure 19. MDS plot of macroinvertebrate data using on log_{10} abundance categories and based on Bray-Curtis Similarity. Samples are coloured by type (top) and season (bottom). Stress was high (0.18).

[9] Although the p-value was significant, the Global R-value of 0.314 indicates the groups are barely separable.

For significant groups, SIMPER analysis was used to determine which species were typical of a group by providing a list of taxa which were found consistently within most samples from a particular group. Average similarity within dry season samples was 48.0%. Species which typified dry season samples included the ephemeroptera *Cloeon* sp., the dytiscid beetles *Tiporus tambreyi* and *Cybister tripunctatus*, and Oligochaeta spp. Wet season samples had an average similarity of 49.6% and were characterised by *Hydra* sp., the corixid *Micronecta sp. A*, and the chironomid *Larsia ?albiceps*.

Table 14. Bray-Curtis similarity matrix for wet season macroinvertebrate samples.

	KB50	KB56	KB57	KB58	Ref59	Ref60	Ref61	JC90	JC91	KBNW1
KB50										
KB56	52.48									
KB57	42.86	58.12								
KB58	58.33	62.89	44.44							
Ref59	39.47	43.36	34.38	47.06						
Ref60	49.64	46.88	35.40	45.16	46.04					
Ref61	45.75	54.17	37.21	46.78	45.16	64.29				
JC90	49.18	35.40	34.69	34.29	38.71	51.38	41.60			
JC91	55.84	57.93	40.00	52.33	42.31	62.41	53.50	44.44		
KBNW1	46.26	47.83	42.28	52.12	32.21	40.30	37.33	31.93	34.44	
KBNW2	50.00	60.74	36.67	54.32	49.32	42.75	46.26	36.21	41.89	41.13

3.5 Fish

3.5.1 Species richness

The fish fauna of the Pilbara is characterised by low species diversity yet high levels of endemicity; over 42% of species recorded from the Pilbara are restricted to the region (Unmack 2001, Allen *et al.* 2002). Masini (1988) found the relatively clear waters of permanent and semi-permanent waterbodies supported the best developed fish assemblages in the region. In a study of the biogeography of Australian fish fauna, Unmack (2001) recognised ten distinct freshwater fish biogeographic provinces, of which the Pilbara Province was one. This region was considered distinct because its fauna did not cluster with other drainages in multivariate (parsimony and UPGMA) analysis of fish distribution patterns (Unmack 2001).

Allen *et al.* (2002) suggested the sparse freshwater fish fauna of the Pilbara was due to its aridity. The fish which inhabit the region are adapted to the extreme conditions and many have strategies for surviving drought (Unmack 2001). For example, Australia's most widespread native fish, the spangled perch (*Leiopotherapon unicolor*), is thought to survive drought by aestivating in wet mud or under moist litter in ephemeral waterbodies (Allen *et al.* 2002). Although conclusive evidence is still required to validate this hypothesis, anecdotal evidence does exist. This species is often found in large numbers shortly after rain in locations which were previously dry and have no connection to permanent water. Spangled perch can migrate in very shallow waters, and can be found in any temporary water of the Pilbara following rainfall, including wheel ruts of vehicle tracks (Allen *et al.* 2002). They are known to tolerate extremes in the aquatic environment (Llewellyn 1973, Beumer 1979, Glover 1982) and occupy a wide range of habitats (Bishop *et al.* 2001, Allen *et al.* 2002). Spangled perch and western rainbowfish are the only species known from an area in the Pilbara with little or no surface run-off in the Great Sandy Desert (Morgan and Gill 2004).

Five of the twelve freshwater fish species known from the Pilbara were recorded during the current study (Table 15). These were the western rainbowfish *Melanotaenia australis*, Hyrtl's tandan (eel-tailed catfish) *Neosiluris hyrtlii* (Plate 5), spangled perch *Leiopotherapon unicolor* (Plate 5), Fortescue grunter *Leiopotherapon aheneus* (Plate 5) and barred grunter *Amniataba percoides*. Spangled perch and western rainbowfish were the most common species recorded. Spangled perch were recorded from all sites, while the barred grunter was only recorded from JC91 (Table 15).

Plate 5. Photos of fish collected from WPIOP waterbodies. Hyrtl's tandan, *Neosiluris hyrtlii* (top), Spangled perch, *Leiopotherapon unicolor* (bottom left), and Fortescue grunter *Leiopotherapon aheneus* (bottom right) (photos by Jess Lynas/WRM).

Table 15. Fish species collected during the current study. ✓ = collected in the dry season *= collected in the wet season 2009.

Type	Site	Western rainbowfish	Hyrtl's tandan	Spangled perch	Fortescue grunter	Barred grunter
	KB50	✓*	✓	✓*	✓*	
	KB56	✓*	✓*	✓*	✓*	
	KB57	✓*	✓	✓*	*	
	KB58	✓*		✓*	*	
	KBNW1	*	*	*		
Potential exposed	**KBNW2**	*		*		
	REF59			✓*		
	REF60			✓*		
	REF61			✓*	✓	
	JC90	✓*	✓*	✓*	✓*	
Reference	**JC91**	✓*		✓*	✓*	*

Generally, the fish recorded from the WPIOP study area are common widespread species. However, the Fortescue grunter has a restricted distribution within the Pilbara Region of Western Australia. It is only known from the Fortescue, Robe and upper Ashburton (Nicholl's Spring) river systems (Allen *et al.* 2002). The Fortescue grunter is reasonably common within its

range. This species is currently listed as 'Lower Risk Near Threatened' on the IUCN Redlist of Threatened Species (IUCN 2009). Its status is considered to require updating (IUCN 2009).

3.5.2 Length-frequency analysis

Breeding characteristics of fish species in the Pilbara, such as fecundity and the size at first maturity, vary between river systems and rainfall zone. Beesley (2006) found life history strategies of fish species in the Fortescue River lay between 'opportunistic' and 'periodic', reflecting the seasonal yet unpredictable nature of rainfall in the region.

Western rainbowfish

Breeding in western rainbowfish (*Melanotaenia australis*) occurs throughout the year, with multiple spawning bouts which take full advantage of the regions intermittent rainfall and streamflow (Beesley 2006). Morgan *et al.* (2002) captured small juveniles on most sampling occasions in the Fitzroy River. The size at first maturity varies between river systems, but western rainbowfish generally attain a maximum size of 110 mm TL (Morgan *et al.* 2002).

The length-frequency plot of western rainbowfish from the Kens Bore sites along Red Hill Creek show a range of size-classes, including new recruits (<30 mm), juveniles, sub-adults and adults (Figure 20). This suggests good recruitment and some degree of population stability, with juveniles and adults through all size classes present in the population. There were a high proportion of new recruits collected from KBNW sites in the wet season of 2009, with few juveniles and no adults collected (Figure 20). The two Jewel Cochrane sites supported western rainbowfish of most size classes, but in much lower abundances than Kens Bore sites (Figure 20).

Hyrtl's tandan (catfish)

Very little is known of the breeding ecology of Hyrtl's tandan (*Neosiluris hyrtlii*). It is thought that individuals may mature in their first year at a size of approximately 135 mm TL for both sexes (Lake 1971, Bishop *et al.* 2001). Species of *Neosilurus* catfish usually attain a maximum size of only 200 mm however, *N. hyrtlii*, along with *N. ater*, can reach up to 400 mm TL (Lake 1971, Bishop *et al.* 2001). Breeding is thought to occur in the early wet season (Morgan *et al.* 2002, Bishop *et al.* 2001). It is at this time when initial flooding increases the area and diversity of aquatic habitat available, while also initiating increases in plankton and other foods (Bishop *et al.* 2001).

Hyrtl's tandan catfish were recorded in much lower abundances than western rainbowfish or spangled perch. Of the small number of catfish collected from the WPIOP study area, a small proportion were likely sexually mature adults (<135 mm; Figure 21). New recruits were collected from Jewel Cochrane sites and KB50 (Figure 21).

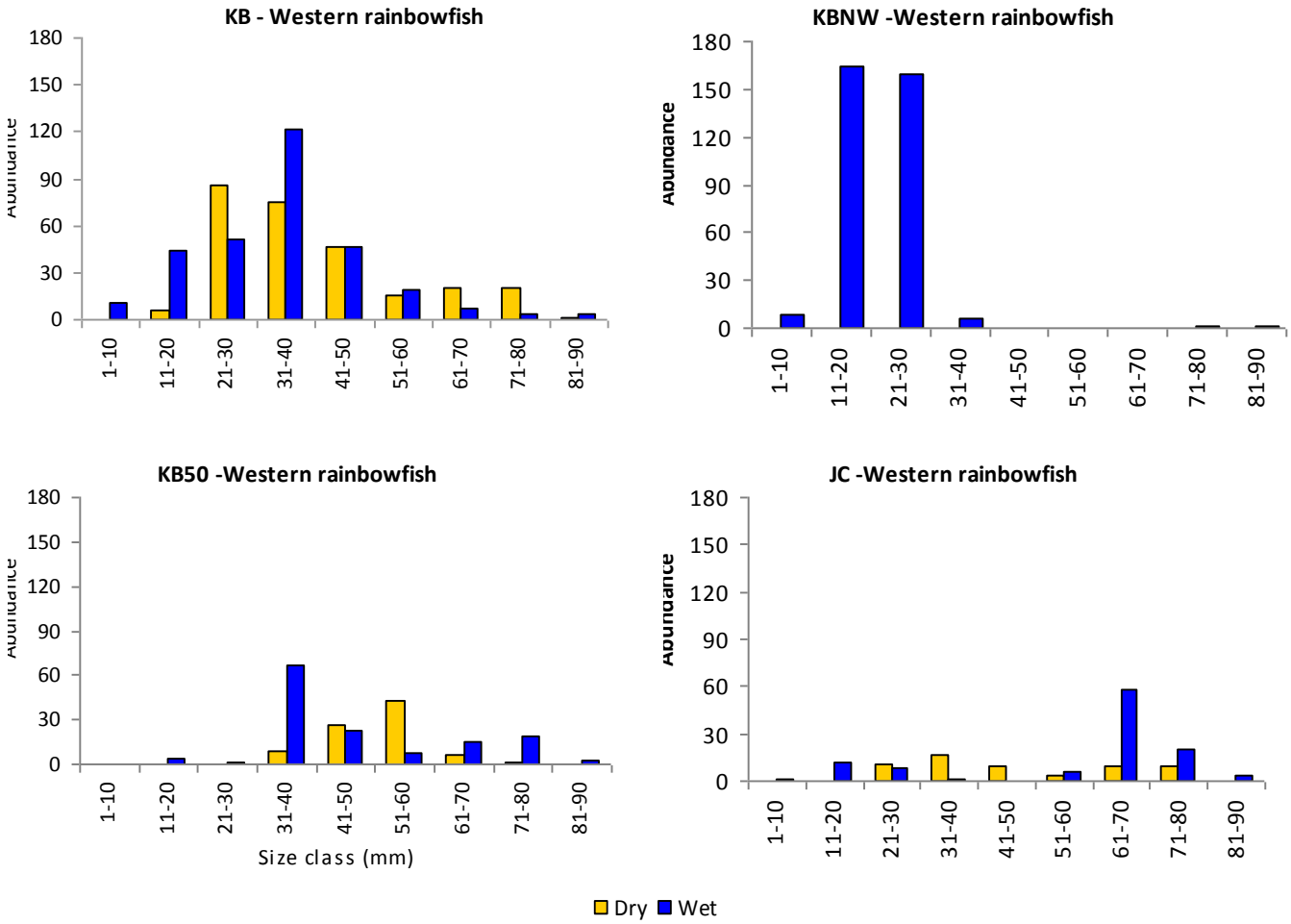


Figure 20. Length-frequency histograms for western rainbowfish recorded from the WPIOP study area during the wet season of 2008 and dry season of 2009.

Spangled perch

Breeding in spangled perch (*Leiopotherapon unicolor*) of the Pilbara occurs during the summer wet season, between late November and March (Beesley 2006, Morgan *et al.* 2002). During this time, multiple spawning events are known to occur (Beesley 2006). In the Fitzroy River, Morgan *et al.* (2002) collected mature specimens in summer and larvae at the end of the wet season, indicating that spawning coincided with the flooding of the river. Spangled perch mature in their first year at approx. 58 mm TL for males and 78 mm TL for females. They reach a maximum size of 300 mm TL. A high abundance of juvenile spangled perch (< 50 mm) were recorded from Kens Bore in the dry season, but none in the wet (Figure 22). Generally, few large adults (> 160 mm) were collected, with the exception of a number from KB50 in the dry season (Figure 22). Spangled perch from Jewel Cochrane included no juveniles, but some sub-adults and adult fish were collected (Figure 22).

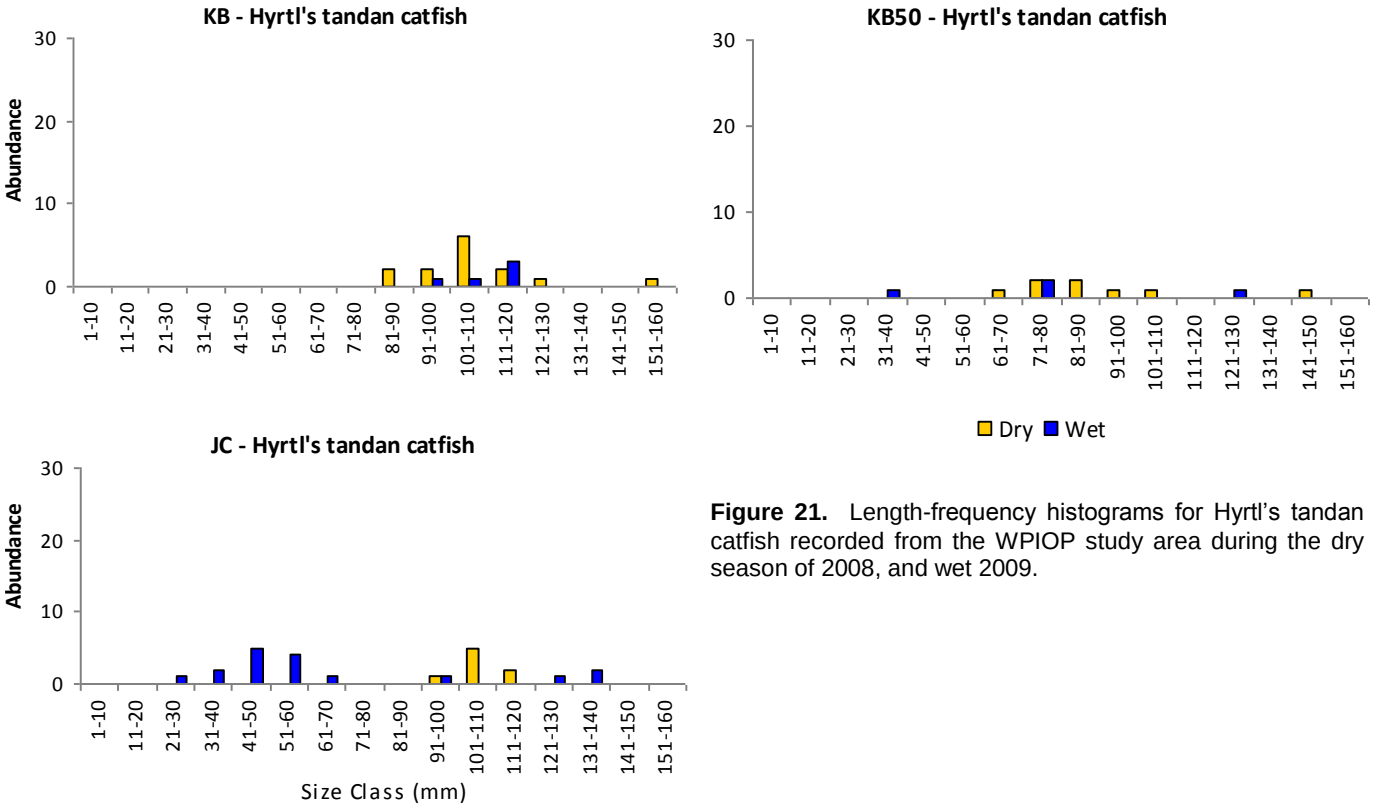


Figure 21. Length-frequency histograms for Hyrtl's tandan catfish recorded from the WPIOP study area during the dry season of 2008, and wet 2009.

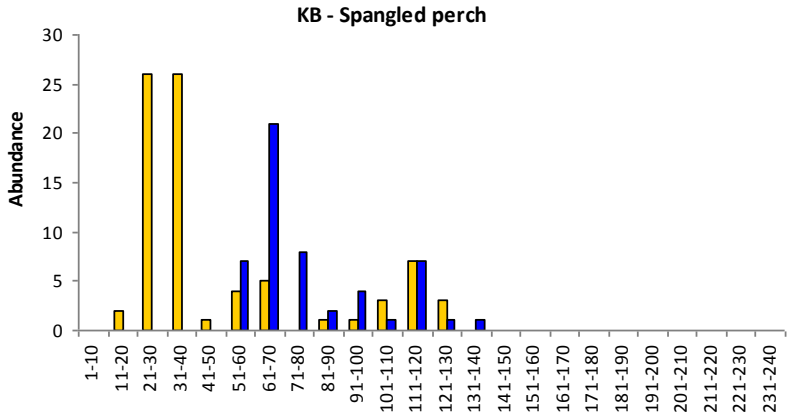
Fortescue grunter

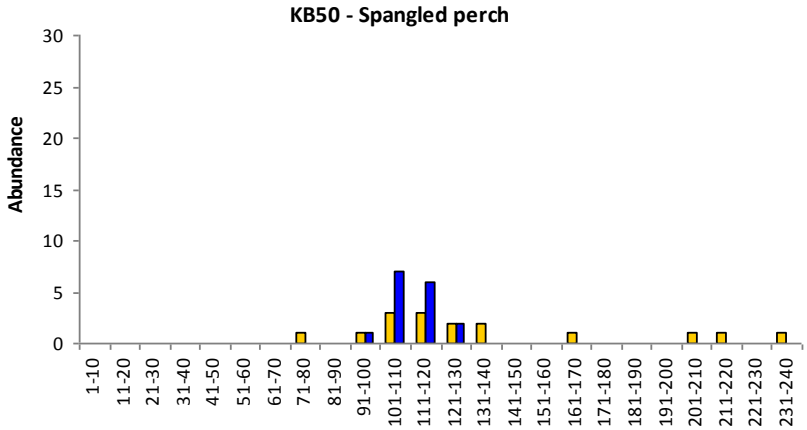
Unfortunately little is known about the biology of the Fortescue grunter. Few specimens were collected from the study area, however Kens Bore sites recorded both small and large-sized individuals (Figure 23). Jewel Cochrane supported Fortescue Grunter within the middle size-classes (Figure 23).

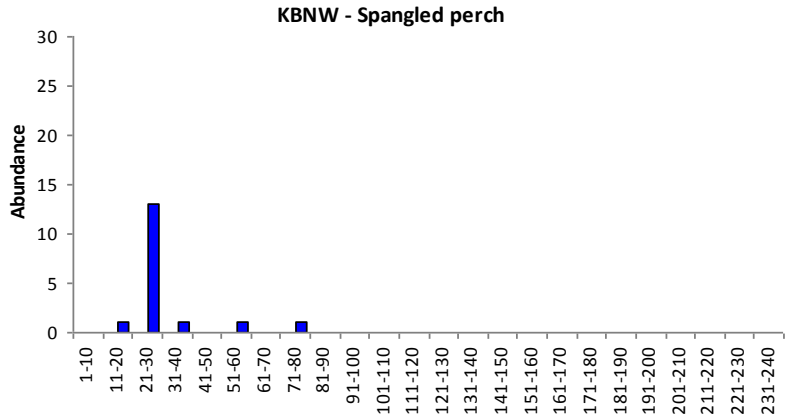
Barred grunter

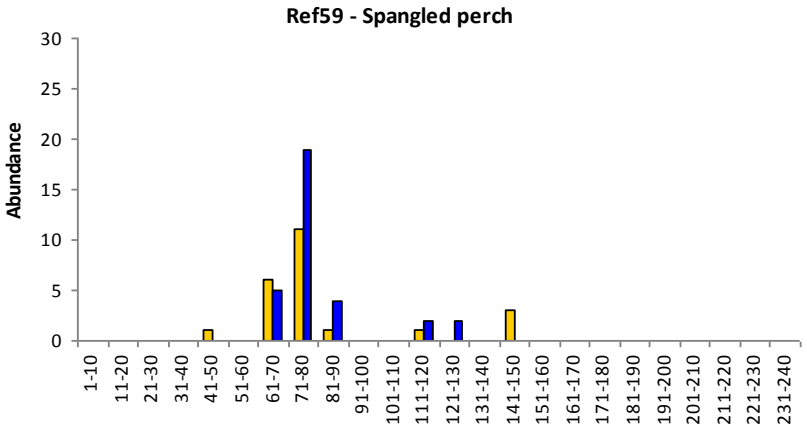
The barred grunter is widely distributed in coastal drainages from the Ashburton River in the Pilbara Region of Western Australia, around northern Australia, south to the Burnett River in Queensland (Allen *et al.* 2002). Breeding is thought to take place between August and March (Allen *et al.* 2002). Bishop *et al.* (2001) reported that barred grunter spawn at the onset of the wet season and grow about 30 mm in six months. Size at first maturity varies between sexes, with males being sexually mature at around 77 mm (SL) and females at 88 mm (Rowland 2001). This species is highly fecund (Allen *et al.* 2002), with females between 70 and 90 g spawning up to 77 000 demersal eggs (Merrick and Schmida 1984, Hebert and Peeters 1995). Barred grunter attains a maximum size of up to 200 mm (Rowland 2001).

Insufficient numbers of barred grunter were recorded during the current study to undertake length-frequency analysis.


KB - Spangled perch
Abundance
0
5
10
15
20
25
30
1-10
11-20
21-30
31-40
41-50
51-60
61-70
71-80
81-90
91-100
101-110
111-120
121-130
131-140
141-150
151-160
161-170
171-180
181-190
191-200
201-210
211-220
221-230
231-240


KB50 - Spangled perch
Abundance
0
5
10
15
20
25
30
1-10
11-20
21-30
31-40
41-50
51-60
61-70
71-80
81-90
91-100
101-110
111-120
121-130
131-140
141-150
151-160
161-170
171-180
181-190
191-200
201-210
211-220
221-230
231-240


KBNW - Spangled perch
Abundance
0
5
10
15
20
25
30
1-10
11-20
21-30
31-40
41-50
51-60
61-70
71-80
81-90
91-100
101-110
111-120
121-130
131-140
141-150
151-160
161-170
171-180
181-190
191-200
201-210
211-220
221-230
231-240


Ref59 - Spangled perch
Abundance
0
5
10
15
20
25
30
1-10
11-20
21-30
31-40
41-50
51-60
61-70
71-80
81-90
91-100
101-110
111-120
121-130
131-140
141-150
151-160
161-170
171-180
181-190
191-200
201-210
211-220
221-230
231-240

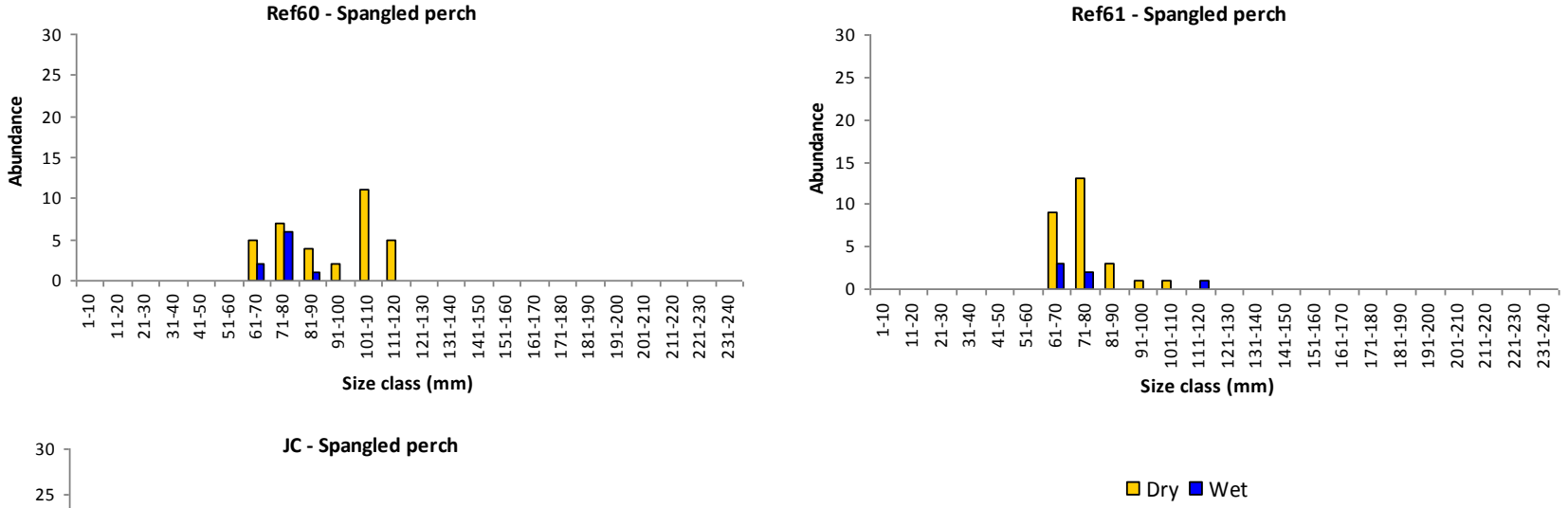


Figure 22. Length-frequency histograms for spangled perch recorded from the WPIOP study area during the dry season of 2008, and wet 2009.

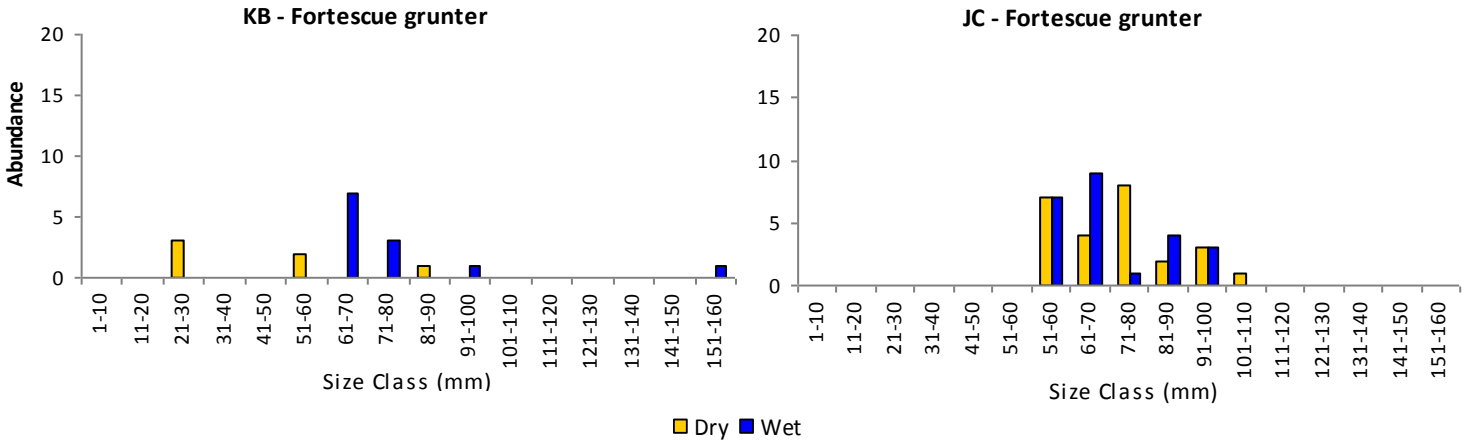


Figure 23. Length-frequency histograms for Fortescue grunters recorded from the WPIOP study area during the wet season of 2008 and dry season of 2009.

4 POTENTIAL ECOLOGICAL IMPACTS

The mine plan for development of the Ken's Bore orebody indicates that de-watering of the developing pit area will be necessary, with potential need to discharge excess water into a creekline (possibly Red Hill Creek). As a result of this dewatering/discharge program, there are potential impacts to the aquatic ecosystem. This may be either as a result of dewatering drawdown and drying of existing aquatic habitats, or discharge of dewatering water to the creekline, resulting in a changed flow regime. The extent of any changes will depend upon the footprint of the dewatering drawdown, and the volume of water to be discharged and the resultant flow regime. Until these parameters are better known, only the generalities of potential impacts to permanent pools and creekline aquatic fauna are discussed

4.1 De-watering impacts

Dewatering of the aquifer will likely result in the drawdown and seasonal drying of permanent pools on Red Hill Creek adjacent to the ore body. Permanent pools in the Pilbara are known to provide important dry season refuges, critical for the survival of aquatic fauna (fish, invertebrates etc), and loss of these pools can result in localised extinctions of species. The footprint of the dewatering drawdown needs to be modelled to provide an indication of the up- and down-stream extent of drawdown, and to determine any permanent pools likely to be affected. Once determined, it will be possible to identify fauna likely to be affected, and their conservation values.

4.2 Discharge operation impacts

4.2.1 De-watering discharge

It is envisaged that water quality of the creeklines will not be adversely impacted by de-watering discharge operations as the quality of the water in the aquifer around the orebody is assumed to be comparable to surface waters. However, a consideration of any mining operation is the exposure of rock types with acid-generating potential, leading to acid mine drainage and associated implications of reduced pH and mobilised dissolved metals. Acidic water (pH $< \sim 5$) may adversely impact aquatic fauna either directly, by interfering with physiology, such as calcium pumps, or indirectly through toxicity of bioavailable metals mobilised by the low pH water. If the orebody contains any pyritic/sulphide rock types in the pit area then there is a potential for ARD and concomitant effects on water quality. Any ARD issues (i.e. increased acidity) would need to be managed on-site prior to discharge of dewatering water, especially as levels of some metals are already naturally elevated. Reduced pH would likely increase the concentrations of bioavailable metals, with potential for metal toxicity to freshwater biota.

Again, depending on the volume of water to be discharged, the infrastructure for discharge needs to be considered. Often dewatering water is discharged to creeklines via a pipe/gabion, with relatively high discharge velocities resulting. The discharge outlet needs to be specifically designed to reduce the impact of high energy flows on the creekline, particularly as the creeklines in the area are predominantly sand/gravel, and therefore easily eroded/mobilised. Ideally, a structure should be built of cobble and boulder-sized rocks leading into an artificial channel before entering the natural creekline. This effectively reduces the energy of the discharge water and thereby reduces potential impacts on the receiving environment, such as erosion and increased turbidity.

Also, depending on volumes to be discharged, there is an option of continuous medium to low discharge, or staged/intermittent high discharge. The former may lead to permanent surface flows for an, as yet, unknown distance, whereas the latter may result in intermittent surface flows. Both options have advantages and disadvantages to the ecology, which need to be considered. For example, the establishment of permanent surface flows for some distance down Red Hill Creek will result in the ecology of the system changing to adapt to the changed hydrology. This will likely result in the loss of species adapted to intermittent flows, with colonisation by species adapted to permanent flows. Changes in riparian vegetation and aquatic plants will likely result, which will ultimately affect food webs and habitat; composition of aquatic insects and crustaceans, and possibly fish and higher vertebrates if present, including terrestrial fauna dependent on the water. In contrast, intermittent releases, with intermittent flows and drying pools, would be more similar to the natural conditions in the creekline; although the periods of surface water may be unseasonal relative to natural hydrology. Delivery of surface flows in the dry season, when the system has stopped flowing and consists of receding pools, will alter the ecology by disrupting seasonal cues (i.e. may break aestivation periods of frogs and turtles) and has the potential to impact populations of species adapted to seasonal flows.

4.2.2 Aquatic invertebrates

Dewatering discharge that results in increased extent and duration of surface flows will increase the "carrying capacity" of the receiving creekline for macroinvertebrates. An increase in habitat heterogeneity and habitat diversity will result from the inundation of riffle zones, submerged macrophyte beds, and overhanging riparian vegetation. Habitat heterogeneity is known to play a crucial role in the structure and trophic organisation of invertebrate communities (Miserendino 2001). Ultimately, an increase in habitat diversity leads to an increase in taxa richness and abundance. Further, the diversity of aquatic fauna is influenced by complex macrophyte habitats due to their high surface area and spatial heterogeneity (Gregg and Rose 1985, Lombardo 1997, Linhart *et al.* 2002). Bella *et al.* (2005) found that, in particular, Coleoptera species richness was driven by macrophyte cover within ponds in Italy. Aquatic vegetation also provides refugia, a food source and breeding sites for macroinvertebrates. Riffle zones are also known to be biodiversity hotspots. A number of studies have found riffles to contain the greatest diversity, biomass and richness of macroinvertebrate fauna (i.e. Brown and Brussock 1991, Barbour *et al.* 1999). The provision of greater areal extent of these habitats, as more of the creek becomes perennial, will result in greater abundance and diversity of macroinvertebrates in the creek.

One perceived negative impact is that alteration to the historic hydrological regime will mean that areas which were once ephemeral will become permanently-flowing. This switch from an ephemeral to permanent freshwater system has implications for fauna specifically adapted to temporary environments. Species which inhabit ephemeral waterbodies must survive in creeks which either dry out completely or are reduced to a series of stagnant pools at certain times of the year. Despite these harsh environmental conditions, many invertebrates are found only in temporary waters (Bunn *et al.* 1989). In fact, Williams (1980) suggests that a number of invertebrates actually require a period of desiccation in order for further development to take place. In addition, biota are specifically adapted to the timing of drying and refilling cycles (Balla 1994). Therefore, any variation in the degree of seasonality can lead to changes in invertebrate community structure (Bunn *et al.* 1989) and changes in life history patterns.

Some species, including those that possess short maturation times, endure the dry season as terrestrial adult stages (*e.g.* mayflies, dragonflies, caddisflies and some beetles). Such species can be known as 'temporary residents' since they must reinvade each time a seasonal waterbody

becomes inundated. Other species possess life history strategies which enable them to remain within a waterbody once surface waters have evaporated. Such taxa can be known as 'permanent residents'. These 'permanent' residents tend to be the microinvertebrates.

There are a number of strategies by which 'permanent residents' can survive in temporary environments. Some species, for example, have drought-resistant spores, eggs or larval stages (e.g. microcrustacea, many species of nematode, some species of simulid, and some species of *Tasmanocoenis* and *Cloeon* mayflies). Of the microinvertebrates, protozoans have cysts, rotifers have ephippia (resting eggs), cladocerans have diapausing eggs, copepods have nauplii (resistant early larval phase) and ostracods have resistant eggs. Most can survive extended periods of drought (Hairston *et al.* 1995). Other species are capable of burrowing into moist sediments of the hyporheic zone, below stones, or into decomposing wood debris (*e.g.*, nemerteans, oligochaetes, Glossiphonid leeches, some species of chironomids, tabanids, and some mayflies). Many bivalves and gastropods are resistant to desiccation and those species which lack an operculum are able to seal their shells with a mucus plug, known as an epiphragm.

Little is known of the habitat and hydrological requirements of permanent resident fauna of Pilbara rivers, however, because these rivers have an unpredictable, episodic flow regime, it is possible the fauna would adapt to the unseasonal intermittently available surface flows and residual pools to complete life cycles and re-enter diapause/drought phase. Unseasonal availability of surface water may pose a problem to fauna adapted to episodic flow, as they may not adapt to the regular availability of water/pools. In particular, this may adversely affect populations in the locality of the dewatering discharge point(s). Discharge over the dry season may lead to early emergence of some species that may then not be able to successfully complete their life-cycle, and so become locally extinct. If dry-season discharge was the preferred de-watering option, then the authors would recommend maintenance of permanent pools year-round to ensure the majority of species had available water for successful recruitment and survival of part of their populations.

It would be expected that fauna would more readily adapt to intermittent discharge of surface water in the wet season, rather than over the dry. Even if this intermittently available surface water disadvantages animals with specific drought-resistant strategies, it should only affect the fauna in the predicted discharge "footprint". It is unlikely that 'permanent' fauna would be lost from the system entirely as these taxa will still be present in the seasonally flowing reaches outside the area of discharge influence, and in adjacent tributary creeks. After mine-closure, these fauna would be expected to re-colonise the reach below the discharge point(s) as a result of natural dispersal following rain events, especially as there is substantial seasonally-flowing catchment upstream of the Ken's Bore project area from which 'permanent' fauna could recolonise the downstream sections post mine closure..

4.2.3 Fish

Provision of permanent flows throughout the survey area will provide additional and greater diversity of habitat for colonisation by fish. Increased flows will increase the carrying capacity of the creek(s) for fish. This will likely result in higher abundances of all fish species that are currently present. The provision of deeper, permanently inundated pools would enable the creeks to support larger fish, thereby increasing fish biomass in the system. In general terms, water depth is an important controlling parameter of habitat availability. It determines the amount of a particular habitat (e.g. woody debris) that is available in the water column, it determines the vertical space in the water column available to mid-water schooling species and it is an important factor in facilitating avoidance of terrestrial predators. Depth can provide

relatively stable, sheltered areas whereas shallow areas are particularly sensitive to reductions in water level, and with increasing depth, light penetration decreases and hence visibility is reduced, again providing protection from predators. Deeper pools support larger fish, and can thereby increase fish biomass in the system. In a study of fish species in the DeGrey River, deeper pools were found to contain significantly larger fish than intermediate and small pools (AW Storey unpub. data). In addition, Bain *et al.* (1988) observed that shallow and slow-flowing areas were used by small, young fish of several species, and deep areas were primarily inhabited by larger, older fish. Schlosser (1982a, b), Finger (1982) and Moyle and Baltz (1985) similarly observed these relationships. Therefore, provision of deeper, permanently inundated pools will result in larger fish in the system.

Under permanent flows, fish would rapidly colonise most reaches and likely establish large populations. This in itself is not a bad scenario, so long as the species are all native species. At mine closure/cessation of dewatering however, any populations established under permanent flows will reduce back to pre-mine levels. Such expansion and contraction is not atypical for the 'boom and bust' ecology of arid systems, with populations expanding in wet seasons and wet years, and contracting under dry seasons and droughts. However, there is a risk of public perception of 'loss of fish populations' post mine closure, especially with a long mine life, and therefore it is very important to thoroughly document conditions prior to development in order to provide a pre-mine ecological condition.

4.2.4 Other aquatic fauna

There is insufficient knowledge of the composition of other water-dependent fauna of the creeks and their water requirements to predict their response to the hydrological scenarios. Therefore, only generic comments are made here on other fauna.

A stable, permanent water source may attract more riparian/terrestrial fauna, such as waterbirds, frogs and lizards. The diversity of such fauna tends to be associated with riparian condition. A combination of a permanent water source and likely increase in riparian vegetation health, complexity and diversity, will potentially increase the abundance and richness of riparian fauna.

Any increase in permanent surface water of good water quality may also act as an attractant for feral animals.

Though not recorded during the current surveys, freshwater turtles are known to be common and widespread throughout the Pilbara and are likely present within the project area. Turtles are known to aestivate in the dry season to avoid desiccation, and are therefore adapted to systems that dry. Intermittent releases in the dry season could adversely affect tortoise by providing triggers for them to break aestivation at the wrong time of year. Generally, turtles require a sufficient period of inundation to feed to a.) provide energy stores to aestivate over the coming dry period, and b.) provide energy to reproduce, with priority given to the former need. If pool duration is too short, then reproduction/fecundity of turtles in the project area may be reduced. A wet season survey targeted at the collection of turtles is required to confirm species composition, conservation significance and wider distribution of turtles. This would involve the use of fyke nets set overnight.

Similarly, many frog species in the Pilbara also aestivate over the dry season to avoid desiccation, emerging following rains to opportunistically mate and lay eggs. Though not recorded during the current surveys, frogs are also likely to frequent pools and water courses in the project area. Like turtles, intermittent releases could adversely affect frog populations by providing triggers for

them to break aestivation at the wrong time of year, affecting reproduction success if pool duration is insufficient for off-spring to reach maturity, and ultimately affecting population sizes. However, provision of permanent flows may be advantageous to some frog species, resulting in increased populations. As with fish, these populations would collapse upon mine closure/cessation of discharges. Implications for frog populations are not known and should be assessed specifically. A wet season survey targeted at the collection of frogs is required to confirm species composition, conservation significance and wider distribution of species. This would involve the collection of tadpoles caught in macroinvertebrate sweeps for identification in the laboratory, as well as identification of species based on male calls heard at the time of sampling.

5 CONCLUSIONS

5.1 Water Quality

The water quality of WPIOP waterbodies was generally good and characterised by circum-neutral pH, high dissolved oxygen levels and fresh waters. However, the dissolved oxygen concentration recorded from KB50 in the dry season was extremely low and the fish fauna were showing obvious signs of stress at the time of sampling (air-breathing). The total nitrogen levels recorded from most sites in the study area were elevated and in excess of the ANZECC/ARMCANZ (2000) trigger value for the protection of aquatic systems. In addition, a number of heavy metals appear to be naturally elevated, including boron, copper and zinc. Multivariate statistics showed no significant difference in water quality between potential exposed and reference sites.

5.2 Microinvertebrates

The microinvertebrate fauna from WPIOP waterbodies was highly diverse with 135 taxa being collected in the dry and wet seasons. These systems appear to support a greater diversity than other Pilbara systems recently sampled (Russ Shiel, Uni of Adelaide, pers. comm.) A significantly greater number of taxa was recorded in the wet than the dry season, which is a pattern regularly observed in the Pilbara. There was no difference in taxa richness between potential exposed and reference sites. Of interest within the microinvertebrate fauna was the collection of a number of taxa endemic to Australia, including the protist *Difflugia australis*, the rotifers *Lecane noobijupi* and *Lecane batillifer*, and the Cladocera *Alona rigidicaudis* and *Moina* cf. *micrura*. Australian endemic species were recorded in low proportions from both potential exposed and reference sites. Furthermore, two microinvertebrate species collected during the current study were classified as new records for either Australia or Western Australia. The collection of the protist *Difflugia capreolata* from the potential exposed site KBNW2 was a new record for Australia. The rotifer *Cephalodella gigantea* collected from the reference site JC91 constituted a new record for Western Australia.

Microinvertebrate assemblages were found to be significantly different between type (potential exposed vs reference sites). The wet season microinvertebrate assemblages of the two Jewel Cochrane sites (reference sites) were more similar to the wet season Kens Bore sites (potential exposed). There was no significant difference in microinvertebrate assemblages between season.

Given that microinvertebrate assemblages are different between potential exposed and reference sites prior to the commencement of mining, relative changes in Bray-Curtis similarities can be used in future monitoring to assess any effect of dewatering and possible discharge operations.

5.3 Hyporheic fauna

Three of the reference sites were not able to be sampled for hyporheic fauna due to the lack of appropriate substrate (gravel) and presence of rocky outcrops (REF59, REF60 and REF61).

Sampling of hyporheic habitats from seven sites in the WPIOP study area in the dry season of 2008, and nine sites in the wet season 2009 revealed the presence of hyporheic fauna. The majority of taxa collected in hyporheic samples however, were classified as stygoxene (69%), i.e. they do not have specialised adaptations for groundwater habitats and were likely surface forms present in the saturated alluviums of the creek bed. Of the 45 taxa collected, 2% were classified

as occasional hyporheos stygophiles, 2% were considered true stygobites, and 5% were considered possible hyporheic fauna. None were considered to be permanent hyporheos stygophiles. Species considered to be hyporheos were the stygobitic amphipod ?Paramelitidae spp., occasional stygophile copepod *Microcyclops varicans*, and possible hyporheic taxa Nematoda spp., and Oligochaeta spp. Hyporheic fauna were present at all sites sampled in both the wet and dry seasons.

Stygobitic amphipods were recorded from a number of sites in the WPIOP study area, including the potential exposed sites KB57, KB58, and KBNW2, and reference site JC91. Although these specimens are still awaiting formal identification they were known to be restricted to groundwater and hyporheic environments because they exhibited a number of characteristics unique to stygofauna, including a lack of pigmentation, eyes reduced or absent, and long appendages. These amphipods may be closely related, or the same species as stygobitic amphipods recorded from other parts of the Pilbara (WRM unpub. data). Genetic analysis of the DNA of these specimens is required to determine their conservation significance, and whether they represent a range extension of a currently known species, or a new species to science.

5.4 Macroinvertebrates

A total of 128 taxa of macroinvertebrates were recorded from the twenty sites sampled (dry and wet seasons). The general composition of macroinvertebrate taxa was typical of freshwater systems throughout the world (Hynes 1970), and was dominated by Insecta (84% of taxa). Although the mean number of macroinvertebrate taxa collected was slightly higher in the wet than the dry season, there was no significant difference in taxa richness between seasons. Nor was there any significant difference in the number of taxa collected between site type (i.e. potential exposed vs reference sites).

The majority of macroinvertebrate taxa recorded were common, ubiquitous species. Of interest however, was the collection of species known only from the Pilbara region of Western Australia. Pilbara endemic species included the Odonate *Nannophlebia injibandi* and the haliplid beetle *Haliplus pilbaraensis*. Both species were collected from the potential exposed site KB58.

Multivariate statistics showed that there was no significant difference between macroinvertebrate assemblages of potential exposed sites vs reference sites. Generally, the within type variation was high, i.e. macroinvertebrate assemblages were highly variable among potential exposed sites, and highly variable amongst reference sites. The absence of a difference in assemblage composition between reference and potentially exposed sites is a good result in terms of long-term monitoring as it will better enable testing for effects of mining. Because the macroinvertebrate assemblages of all sites are currently similar / indistinguishable, any future separation of exposed sites from reference sites would indicate a potential mine-related effect.

There was a significant difference in macroinvertebrate assemblages between season, and therefore future monitoring should include wet and dry season sampling.

5.5 Fish

Five of the twelve freshwater fish species known from the Pilbara were recorded during the current study. These were the western rainbowfish *Melanotaenia australis*, Hyrtl's tandan (eel-tailed

catfish) *Neosiluris hyrtlii*, spangled perch *Leiopotherapon unicolor*, Fortescue grunter *Leiopotherapon aheneus* and barred grunter *Amniataba percoides*. Spangled perch and western rainbowfish were the most common species recorded. Spangled perch were recorded from all sites, while the barred grunter was only recorded from JC91.

Generally, the fish recorded from the WPIOP study area are common widespread species. However, the Fortescue grunter has a restricted distribution within the Pilbara Region of Western Australia. It is only known from the Fortescue, Robe and upper Ashburton (Nicholl's Spring) river systems (Allen *et al.* 2002). The Fortescue grunter is reasonably common within its range. This species is currently listed as 'Lower Risk Near Threatened' on the IUCN Redlist of Threatened Species (IUCN 2009). Its status is considered to require updating (IUCN 2009).

6 RECOMMENDATIONS

It is suggested that the current monitoring program, as reported here, be continued on a bi-annual basis to assess the natural spatial and temporal variability of the aquatic ecosystems. The creeks should be sampled in the late wet (March/April) and again in the late dry (Sept/Oct) each year, prior to dewatering and mine construction, in order to provide a baseline. ANZECC/ARMCANZ (2000) recommend that at least three years baseline data are required to establish local trigger levels for water quality, with similar data requirements for assessing changes in aquatic fauna. During project development and dewatering, aquatic monitoring should continue to assess the extent of impacts (if any). It is recommended that a minimum additional two-years baseline data be collected prior to commencement of dewatering in order to separate any effects of the discharge from natural spatial/temporal variability. This is particularly important in view of the presence of the Pilbara endemic species; the odonate *Nannophlebia injibandi* and the haliplid beetle *Haliplus pilbaraensis*

Additional specific recommendations are provided below:

1. Water quality – if ARD is a potential issue, then monitoring pH levels is critical for detecting potential ARD. However, measuring pH alone is probably not adequate because by the time a decline in pH is detected, there is already a problem. ARD has implications for aquatic fauna from the low pH as a stressor, as well as the mobilisation and toxicity of metals. Therefore, it is suggested that a broad range of metals continue to be measured in conjunction with sampling to set a baseline. These data could also be used to develop system-specific guidelines as recommended by ANZECC/ARMCANZ (2000).

2. Water quality – dewatering will reduce residency time of water in the aquifer and this in turn may alter water chemistry. It is therefore suggested that water chemistry at the discharge point(s) be monitored and compared with that from the pit and pre-dewatering data from the aquifer to determine whether there are any differences in ionic composition.

3. Other aquatic biota (turtles and frogs) – wet season sampling should include targeted sampling for turtles and frogs. This would include the use of fyke nets for turtles, the collection of tadpoles caught in macroinvertebrate sweeps for identification in the laboratory, and identification of frogs based on male calls heard at the time of sampling and identifications made from photographs of adults observed in the field.

4. Long-term monitoring – additional monitoring sites may need to be added if dewatering discharge results in a longitudinal extension in surface flows beyond the downstream extent of sites already established. This could either involve establishment of additional sites, or relocating existing sites progressively downstream.

5. The relationship between the local aquifer supporting permanent pools, and the aquifer to be dewatered under the pit is not known. As a potential exists for pit dewatering to result in drawdown of these permanent pools, which support noted ecological values, depth gauges/gauge boards should be established in each pool, calibrated to the deepest point, and monitored at least monthly to detect any abnormal drawdown that may be as a result of pit dewatering. If this should occur, water levels of the pools would need to be supplemented.

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APPENDICES

Appendix 1. Site photographs

POTENTIAL EXPOSED SITES

DRY **WET**

KB50

KB56

KB57

KB58

KBNW1

KBNW2

REFERENCE SITES

DRY **WET**

REF59

REF60

REF61


JC90


JC91

Appendix 2. ANZECC/ARMCANZ (2000) trigger values for the protection of aquatic systems in tropical northern Australia

Table A2-1. Default trigger values for some physical and chemical stressors for tropical Australia for slightly disturbed ecosystems (TP = total phosphorus; FRP = filterable reactive phosphorus; TN = total nitrogen; NOx = total nitrates/nitrites; NH4+ = ammonium). Data derived from trigger values supplied by Australian states and territories, for the Northern Territory and regions north of Carnarvon in the west and Rockhampton in the east (ANZECC/ARMCANZ 2000).

	TP ($\mu g\ L^{-1}$)	FRP ($\mu g\ L^{-1}$)	TN ($\mu g\ L^{-1}$)	NOx ($\mu g\ L^{-1}$)	NH_4^+ ($\mu g\ L^{-1}$)	DO % saturation[f]	pH
Aquatic Ecosystem							
Upland River[e]	10	5	150	30	6	90-120	6.0-7.5
Lowland River[e]	10	4	200-300[h]	10[b]	10	85-120	6.0-8.0
Lakes & Reservoirs	10	5	350[c]	10[b]	10	90-120	6.0-8.0
Wetlands[3]	10-50[g]	5-25[g]	350-1200[g]	10	10	90[b]-120 [b]	6.0-8.0

b = Northern Territory values are 5µgL^{-1} for NO_x, and <80 (lower limit) and >110% saturation (upper limit) for DO;
c = this value represents turbid lakes only. Clear lakes have much lower values;
e = no data available for tropical WA estuaries or rivers. A precautionary approach should be adopted when applying default trigger values to these systems;
f = dissolved oxygen values were derived from daytime measurements. Dissolved oxygen concentrations may vary diurnally and with depth. Monitoring programs should assess this potential variability;
g = higher values are indicative of tropical WA river pools;
h = lower values from rivers draining rainforest catchments.

Table A2-2. Default trigger values for salinity and turbidity for the protection of aquatic ecosystems, applicable to tropical systems in Australia (ANZECC/ARMCANZ 2000).

Aquatic Ecosystem		**Comments**
Salinity	**(µs/cm)**	
Upland & lowland rivers	20-250	Conductivity in upland streams will vary depending on catchment geology. The fist flush may result in temporarily high values
Lakes, reservoirs & wetlands	90-900	Higher conductivities will occur during summer when water levels are reduced due to evaporation
Turbidity	**(NTU)**	
Aquatic Ecosystem		
Upland & lowland rivers	2-15	Can depend on degree of catchment modification and seasonal rainfall runoff
Lakes, reservoirs & wetlands	2-200	Most deep lakes have low turbidity. However, shallow lakes have higher turbidity naturally due to wind-induced re-suspension of sediments. Wetlands vary greatly in turbidity depending on the general condition of the catchment, recent flow events and the water level in the wetland.

Table A2-3. Trigger values for toxicants at alternative levels of protection.

Chemical		Trigger values for freshwater (μgL^{-1})				Trigger values for marine water (μgL^{-1})			
		Level of protection (% species)				Level of protection (% species)			
		99%	95%	90%	80%	99%	95%	90%	80%
METALS & METALLOIDS									
Aluminium	pH >6.5	27	55	80	150	ID	ID	ID	ID
Aluminium	pH <6.5	ID	ID	ID	ID	ID	ID	ID	ID
Antimony		ID	ID	ID	ID	ID	ID	ID	ID
Arsenic (As III)		1	24	94 [C]	360 [C]	ID	ID	ID	ID
Arsenic (AsV)		0.8	13	42	140 [C]	ID	ID	ID	ID
Beryllium		ID	ID	ID	ID	ID	ID	ID	ID
Bismuth		ID	ID	ID	ID	ID	ID	ID	ID
Boron		90	370 [C]	680 [C]	1300 [C]	ID	ID	ID	ID
Cadmium	H	0.06	0.2	0.4	0.8 [C]	0.7 [B]	5.5 [B, C]	14 [B, C]	36 [B, A]
Chromium (Cr III)	H	ID	ID	ID	ID	7.7	27.4	48.6	90.6
Chromium (CrVI)		0.01	1.0 [C]	6 [A]	40 [A]	0.14	4.4	20 [C]	85 [C]
Cobalt		ID	ID	ID	ID	0.005	1	14	150 [C]
Copper	H	1.0	1.4	1.8 [C]	2.5 [C]	0.3	1.3	3 [C]	8 [A]
Gallium		ID	ID	ID	ID	ID	ID	ID	ID
Iron		ID	ID	ID	ID	ID	ID	ID	ID
Lanthanum		ID	ID	ID	ID	ID	ID	ID	ID
Lead	H	1.0	3.4	5.6	9.4 [C]	2.2	4.4	6.6 [C]	12 [C]
Manganese		1200	1900[C]	2500[C]	3600[C]	ID	ID	ID	ID
Mercury (inorganic)	B	0.06	0.6	1.9 [C]	5.4 [A]	0.1	0.4 [C]	0.7 [C]	1.4 [C]
Mercury (methyl)		ID	ID	ID	ID	ID	ID	ID	ID
Molybdenum		ID	ID	ID	ID	ID	ID	ID	ID
Nickel	H	8	11	13	17 [C]	7	70 [C]	200 [A]	560[A]
Selenium (Total)	B	5	11	18	34	ID	ID	ID	ID
Selenium (SeIV)	B	ID	ID	ID	ID	ID	ID	ID	ID
Silver		0.02	0.05	0.1	0.2 [C]	0.8	1.4	1.8	2.6 [C]
Thallium		ID	ID	ID	ID	ID	ID	ID	ID
Tin (inorganic, SnIV)		ID	ID	ID	ID	ID	ID	ID	ID
Tributyltin (as μg/L Sn)		ID	ID	ID	ID	0.0004	0.006 [C]	0.02 [C]	0.05 [C]
Uranium		ID	ID	ID	ID	ID	ID	ID	ID
Vanadium		ID	ID	ID	ID	50	100	160	280
Zinc	H	2.4	8.0 [C]	15 [C]	31 [C]	7	15 [C]	23 [C]	43 [C]
NON-METALLIC INORGANICS									
Ammonia	D	320	900 [C]	1430 [C]	2300 [A]	500	910	1200	1700
Chlorine	E	0.4	3	6 [A]	13 [A]	ID	ID	ID	ID
Cyanide	F	4	7	11	18	2	4	7	14
Nitrate	J	17	700	3400 [C]	17000 [A]	ID	ID	ID	ID
Hydrogen sulfide	G	0.5	1.0	1.5	2.6	ID	ID	ID	ID

Appendix 3. Abundance of microinvertebrates from each site sampled. Values are log_{10} abundance classes where 1 = 1 individual, 2 = 2-10 individuals, 3 = 10 – 100, and so on.

Table A3-1: Dry season 2008.

			Potential exposed				Reference				
			KB50	KB56	KB57	KB58	Ref59	Ref60	Ref61	JC90	JC91
PROTISTA											
Ciliophora											
		Didinium sp.	0	0	0	1	0	0	0	0	0
	Euplotiidae	*Euplotes*	3	0	0	3	0	0	0	0	0
Rhizopoda											
	Arcellidae	*Arcella discoides*	1	1	2	1	1	1	1	3	2
		Arcella megastoma	0	0	0	1	0	0	0	0	0
		domed *Arcella* [med.]	0	1	0	1	0	0	0	0	0
		discoid *Arcella* [sm]	0	0	1	0	0	0	0	1	0
	Centropyxidae	*Centropyxis ecornis*	0	1	0	2	1	0	0	0	0
		cf. *Cyclopyxis*	0	0	0	0	1	0	0	0	0
	Difflugiidae	*Difflugia* cf. *australis*	0	1	0	1	0	0	0	0	0
		Difflugia [med, glob, dk br]	0	1	1	0	0	0	0	0	0
	Euglyphidae	*Euglypha* a	0	0	1	0	0	0	0	0	0
	Lesquereusiidae	*Lesquereusia spiralis*	0	0	2	1	1	0	0	0	0
		Netzelia tuberculata	1	0	0	0	0	0	0	0	0
ROTIFERA											
Bdelloidea											
		indet v sm bdelloid [X main unci]	0	0	2	0	0	0	0	0	0
		indet sm bdelloid [2 main unci]	1	1	3	0	0	0	1	0	0
		indet lge bdelloid [2 main unci]	0	0	2	4	0	0	0	2	0
	Atrochidae	*Cupelopagis vorax*	0	1	0	0	0	0	0	0	0
Monogononta											
	Brachionidae	*Brachionus angularis*	2	0	0	2	2	0	0	0	0
		Brachionus budapestinensis	2	1	1	3	0	0	0	0	0
		Brachionus dichotomus	0	0	0	0	0	0	1	0	0
		Brachionus quadridentatus ancylognathus	0	0	0	1	0	0	0	0	0
		Brachionus cf. *urceolaris*	0	1	0	0	0	0	0	0	0
		Brachionus	0	0	0	0	0	0	0	0	1
		Keratella cochlearis s.l.	0	0	0	0	4	3	4	0	0
		Keratella procurva	3	4	0	0	3	0	2	0	0

			Potential exposed				Reference				
			KB50	KB56	KB57	KB58	Ref59	Ref60	Ref61	JC90	JC91
	Conochilidae	*Conochilus dossuarius*	4	2	0	0	0	0	0	0	0
	Epiphanidae	*Microcodides chlaena*	0	0	2	0	0	0	0	0	0
	Hexarthridae	*Hexarthra intermedia*	2	0	0	0	3	0	0	0	0
	Lecanidae	*Lecane arcula*	0	0	2	0	0	0	0	0	0
		Lecane batillifer	0	0	2	0	0	0	0	0	0
		Lecane bulla	1	0	3	0	1	0	1	0	0
		Lecane leontina	0	0	0	0	0	0	1	0	0
		Lecane ludwigii	0	0	2	0	0	0	0	0	0
		Lecane noobijupi	0	0	1	0	0	0	0	0	0
		Lecane papuana	0	0	3	0	0	0	0	0	0
		Lecane quadridentata	1	0	0	0	0	0	2	0	0
		Lecane signifera	0	0	1	0	0	0	0	0	0
		Lecane cf. *stenroosi*	0	0	1	0	0	0	0	0	0
		Lecane subtilis	0	0	2	0	0	0	0	0	0
		Lecane (*M.*) a	0	0	0	0	0	0	0	1	0
		Lecane (*M.*) c	0	1	0	0	0	0	0	0	1
	Lepadellidae	*Colurella uncinata bicuspidata*	0	0	2	0	0	0	0	0	0
		Lepadella cf. *patella*	0	0	1	0	0	0	0	0	0
		Lepadella rhomboides	0	0	0	1	0	0	0	0	0
		Lepadella	0	0	0	0	1	0	0	1	0
	Notommatidae	*Cephalodella* cf. *tinca*	2	0	0	0	0	0	0	0	0
		Notommata cf. *tripus*	0	0	2	0	0	0	0	0	0
	Synchaetidae	*Polyarthra dolichoptera*	4	3	0	0	1	0	0	0	0
	Trichocercidae	*Trichocerca pusilla*	0	0	0	0	0	0	0	0	3
		Trichocerca similis	0	0	0	0	0	0	0	0	3
	Trichosphaeridae	*Filinia*	0	0	0	0	1	0	0	0	0
		indet rotifer a [sm]	0	0	1	2	0	0	0	0	0
		indet rotifer b [m]	0	0	0	0	0	0	0	0	0
COPEPODA											
Calanoida											
		Eodiaptomus lumholtzi	0	0	0	0	2	0	3	0	0
		indet calanoid [?*Calamoecia*]	0	0	0	0	0	0	0	1	0
		calanoid copepodites	2	0	0	0	2	0	3	0	0
		calanoid nauplii	0	0	0	0	3	0	4	0	0
Cyclopoida											
		Mesocyclops	0	1	0	0	0	0	0	1	0

			Potential exposed				Reference				
			KB50	KB56	KB57	KB58	Ref59	Ref60	Ref61	JC90	JC91
		Thermocyclops	0	0	0	0	1	0	1	0	0
		Tropocyclops	0	1	0	0	0	0	1	2	0
		cyclopoid copepodites	1	1	0	0	3	3	2	3	3
		cyclopoid nauplii	2	3	2	2	4	4	3	4	3
Harpacticoida											
		indet. Harpacticoid	0	0	0	0	1	0	0	0	0
CLADOCERA											
	Chydoridae	*Chydorus*	0	0	2	0	0	0	0	0	0
	Daphniidae	*Ceriodaphnia cornuta*	0	0	0	0	1	0	1	0	0
	Moinidae	*Moina* cf. *micrura*	0	0	0	0	2	0	1	0	0
	Sididae	*Diaphanosoma*	0	0	0	0	2	0	0	0	0
OSTRACOD											
		Cypretta	0	0	0	0	0	0	1	0	0
		indet. Juv ostracod	0	0	0	0	1	0	1	0	0
		Taxa richness	**16**	**17**	**25**	**15**	**23**	**4**	**19**	**10**	**7**

Table A3-2: Wet season 2009.

			Potential exposed						Reference				
			KB50	KB56	KB57	KB58	KBNW1	KBNW2	Ref59	Ref60	Ref61	JC90	JC91
PROTISTA													
	Euplotiidae	*Euplotes*	0	0	0	0	2	1	0	0	0	0	2
Rhizopoda													
	Arcellidae	*Arcella bathystoma*	0	2	2	0	0	0	0	2	0	0	1
		Arcella discoides	1	1	2	2	2	2	0	0	0	2	2
		Arcella hemisphaerica	4	3	0	2	0	1	0	0	0	0	0
		Arcella megastoma	0	0	0	2	0	0	0	0	0	0	0
		Arcella mitrata	0	2	0	0	0	0	0	0	0	0	0
		domed *Arcella* [med.]	2	3	3	3	1	3	0	3	0	0	0
		discoid *Arcella* [sm]	0	0	0	0	3	3	0	0	0	0	0
	Centropyxidae	*Centropyxis aculeata*	1	0	0	0	1	0	0	0	0	0	2
		Centropyxis ecornis	1	1	3	1	1	1	0	0	0	0	2
		Centropyxis [med.]	0	2	2	1	1	1	0	0	1	2	2

			Potential exposed						Reference				
			KB50	KB56	KB57	KB58	KBNW1	KBNW2	Ref59	Ref60	Ref61	JC90	JC91
		Centropyxis [tiny]	0	0	2	1	1	0	1	0	0	0	0
	Difflugiidae	*Cucurbitella* cf. *australica*	0	0	0	0	0	0	0	1	0	0	0
		Difflugia cf. *australis*	0	1	2	1	1	0	0	2	0	0	0
		Difflugia capreolata **NR for Aust.**	0	0	0	0	0	1	0	0	0	0	0
		Difflugia corona	0	0	0	0	0	3	0	0	0	0	0
		Difflugia gramen	0	0	2	0	0	0	0	0	0	0	0
		Difflugia limnetica	0	1	0	0	0	0	0	0	0	0	0
		Difflugia [med, glob, dk br]	1	2	0	2	0	0	0	0	1	0	2
		Difflugia [sm, glob, dk br]	0	0	1	0	0	0	0	0	0	0	2
		Difflugia [med, pyriform]	1	2	2	2	2	1	0	0	0	1	3
	Euglyphidae	*Euglypha* a	1	1	2	2	0	0	0	0	0	0	0
		Euglypha b	0	1	0	0	2	0	2	2	0	0	0
	Lesquereusiidae	*Lesquereusia spiralis*	1	3	3	2	1	3	2	1	0	2	2
		Netzelia tuberculata	1	0	3	2	0	4	0	0	0	2	0
ROTIFERA													
Bdelloidea													
		Habrotrocha sp.	0	0	1	0	0	0	0	0	0	0	0
		Philodina cf. *alata*	0	0	0	0	0	0	0	0	0	2	1
		Rotaria sp.	0	0	1	0	0	0	0	0	0	0	0
		indet v sm bdelloid [X main unci]	3	0	2	0	0	0	0	0	0	0	0
		indet sm bdelloid [2 main unci]	1	2	2	3	3	1	0	0	0	3	3
		indet lge bdelloid [2 main unci]	0	1	0	2	2	0	0	0	0	0	2
	Atrochidae	*Cupelopagis vorax*	0	0	0	0	0	0	0	0	0	0	1
Monogononta													
	Brachionidae	*Brachionus angularis*	0	1	0	0	2	0	0	0	0	0	0
		Brachionus dichotomus	0	0	0	0	0	0	0	0	3	0	0
		Keratella cochlearis s.l.	0	0	0	1	0	0	0	3	0	0	0
		Keratella procurva	1	0	0	0	0	0	3	0	0	0	0
		Platyias quadricornis	1	0	0	0	0	0	0	0	0	0	1
	Conochilidae	*Conochilus dossuarius*	0	0	0	0	0	0	0	1	0	0	0
	Dicranophoridae	*Dicranophorus*	1	0	0	1	0	0	0	0	0	1	0
	Epiphanidae	*Microcodides chlaena*	0	0	0	0	2	0	0	0	0	0	0
	Euchlanidae	*Dipleuchlanis propatula*	0	0	0	0	2	0	0	0	0	2	1
		Euchlanis dilatata	0	0	0	0	0	0	1	0	0	0	0
		Euchlanis	0	1	0	2	2	2	0	0	0	0	0
		Tripleuchlanis plicata	0	0	0	0	3	0	0	0	0	2	2
	Gastropodidae	*Ascomorpha*	0	2	0	2	0	0	0	0	0	0	0
	Hexarthridae	*Hexarthra intermedia*	0	0	0	0	0	0	0	0	3	0	0

		Potential exposed						Reference				
		KB50	KB56	KB57	KB58	KBNW1	KBNW2	Ref59	Ref60	Ref61	JC90	JC91
Lecanidae	*Lecane arcula*	1	2	2	0	0	0	0	0	0	0	0
	Lecane batillifer	0	0	2	0	0	0	0	0	0	0	0
	Lecane bulla	2	2	1	3	3	0	0	0	0	3	2
	Lecane cf. *closterocerca*	0	3	0	0	0	0	0	0	0	2	3
	Lecane crepida	0	0	1	0	0	0	0	0	0	2	3
	Lecane curvicornis	2	0	1	2	0	0	0	0	0	0	2
	Lecane elsa	0	0	0	1	0	0	0	0	0	0	0
	Lecane hamata	0	2	0	0	0	0	0	0	0	2	1
	Lecane cf. *hornemanni*	0	0	0	0	0	0	0	0	0	0	1
	Lecane leontina	0	0	0	1	0	0	0	0	0	0	0
	Lecane ludwigii	0	0	3	0	2	1	0	0	0	0	0
	Lecane luna	0	2	1	0	2	0	0	0	0	2	1
	Lecane lunaris	0	0	0	2	0	0	0	0	0	0	0
	Lecane papuana	1	6	0	0	0	0	0	0	0	3	3
	Lecane scutata	0	0	1	0	0	0	0	0	0	0	0
	Lecane signifera	0	0	2	2	0	0	0	0	0	2	1
	Lecane subtilis	0	0	3	0	0	0	0	0	0	0	0
	Lecane ungulata	0	0	0	0	0	0	0	0	0	3	0
	Lecane unguitata	0	0	0	0	0	0	0	0	0	1	1
	Lecane s. str.	0	0	0	0	1	1	0	0	0	0	1
	Lecane (*M.*) a	0	1	0	2	0	0	0	0	0	0	2
	Lecane (*M.*) b	0	2	0	2	0	0	0	0	0	0	1
Lepadellidae	*Colurella uncinata bicuspidata*	0	0	1	0	0	0	0	0	0	1	0
	Colurella sp.	0	0	0	0	2	0	0	0	0	0	0
	Lepadella (*H.*) *cyrtopus*	1	0	1	0	0	0	0	0	0	0	0
	Lepadella (*H.*) *ehrenbergii*	0	0	0	1	0	0	0	0	0	0	2
	Lepadella cf. *biloba*	0	0	1	0	0	0	0	0	0	0	0
	Lepadella rhomboides	0	0	0	0	0	0	0	0	0	3	0
	Lepadella triptera	0	0	0	0	0	0	0	0	0	3	2
	Lepadella	0	0	0	1	2	0	0	0	0	2	1
Lindiidae	*Lindia truncata*	0	0	1	0	0	0	0	0	0	0	0
Notommatidae	*Cephalodella forficula*	0	0	2	0	0	1	0	0	0	0	3
	Cephalodella cf. *gibba*	0	2	0	0	0	0	0	0	0	0	0
	Cephalodella gigantea **NR for WA**	0	0	0	0	0	0	0	0	0	0	1
	Cephalodella sp.	0	0	0	0	1	0	0	0	0	0	0
	cf. *Eosphora*	0	0	0	0	0	0	0	0	0	2	0
	Notommata cf. *tripus*	2	0	0	2	0	1	0	0	0	3	0

			Potential exposed						Reference				
			KB50	KB56	KB57	KB58	KBNW1	KBNW2	Ref59	Ref60	Ref61	JC90	JC91
		notommatid spp.	0	2	0	0	0	0	0	0	0	0	0
		cf. *Resticula*	0	0	0	0	0	0	0	0	0	2	0
	Synchaetidae	*Polyarthra dolichoptera*	0	0	0	0	0	0	0	0	0	0	2
		cf. *Synchaeta*	0	0	0	0	0	0	0	2	0	0	0
	Testudinellidae	*Testudinella patina*	1	1	1	2	0	0	0	0	0	0	1
	Trichotriidae	*Macrochaetus* cf. *altamirai*	0	0	1	0	0	0	0	0	0	0	2
	Trichocercidae	*Trichocerca* a	0	2	0	0	0	0	0	0	0	0	0
		Trichocerca b	0	1	0	0	0	0	0	0	0	0	0
	Trichosphaeridae	indet rotifer a [sm]	1	2	2	0	2	0	0	0	2	0	3
		indet rotifer b [m]	0	0	0	0	0	0	0	0	0	0	2
COPEPODA													
Calanoida													
		Eodiaptomus lumholtzi	0	0	0	0	0	0	0	1	2	0	0
		indet calanoid [?*Calamoecia*]	0	0	0	0	1	0	2	0	0	0	0
		calanoid copepodites	0	0	0	0	0	0	2	3	2	0	0
		calanoid nauplii	0	0	0	0	0	0	2	3	3	0	0
Cyclopoida													
		Mesocyclops	0	0	0	0	0	0	2	1	0	1	1
		Microcyclops	0	0	0	1	0	0	0	0	0	0	0
		Thermocyclops	0	0	0	0	0	0	1	1	1	0	0
		Tropocyclops	0	0	0	0	0	0	1	1	1	0	0
		cyclopoid copepodites	0	0	0	1	1	0	4	4	3	2	2
		cyclopoid nauplii	3	2	2	2	2	2	3	4	4	3	3
CLADOCERA													
	Chydoridae	*Alona rectangula novaezealandiae*	0	0	0	0	0	1	0	0	0	0	0
		Alona rigidicaudis	0	0	0	0	0	0	0	0	0	0	1
		Alonella clathratula	0	0	0	0	0	0	2	0	0	0	0
		Chydorus	0	2	2	2	0	0	0	0	0	1	0
		Ephemeroporus barroisi	0	0	0	0	0	2	0	0	0	0	1
		Karualona karua	0	0	1	0	0	0	0	0	0	0	0
		juv. chydorid	0	0	0	1	1	0	0	0	0	0	0
	Daphniidae	*Ceriodaphnia cornuta*	0	0	0	0	0	1	2	0	0	0	0
	Macrotrichidae	*Macrothrix* juv.	0	2	1	1	0	1	1	0	0	0	0
	Moinidae	*Moina* cf. *micrura*	0	0	0	0	0	0	1	1	1	0	0
	Sididae	*Diaphanosoma*	0	0	0	0	0	0	1	0	2	0	0

		Potential exposed						Reference				
		KB50	**KB56**	**KB57**	**KB58**	**KBNW1**	**KBNW2**	**Ref59**	**Ref60**	**Ref61**	**JC90**	**JC91**
OSTRACOD												
	Cypretta	1	0	1	0	0	1	1	0	0	0	1
	Limnocythere	1	0	2	1	0	0	0	1	0	0	1
	Stenocypris malcolmsii	0	0	0	0	0	0	1	0	0	0	0
	Stenocypris	0	0	1	1	0	0	0	0	0	0	1
	indet. Juv ostracod	1	3	3	2	1	1	0	1	2	2	2
	Taxa richness	**27**	**37**	**43**	**40**	**32**	**25**	**20**	**20**	**15**	**31**	**49**

Appendix 4. List of taxa collected in hyporheic samples (actual abundance), along with their hyporheic classification category.

Hyporheic classification category:
X = Stygoxene
O = Occasional stygophile
S = Stygobite
PS = Permanent hyporheic stygophile
P = Possible hyporheic
U = Unknown

Table A4-1: Dry season 2008.

				Potential exposed				Reference	
			Cat	KB50	KB56	KB57	KB58	JC90	JC91
GASTROPODA	**Planorbidae**	*Gyraulus hesperus*	**X**	0	0	0	0	1	0
CRUSTACEA									
Ostracoda									
		sp b cf *Candonocypris*	**U**	1	0	0	0	0	0
Copepoda									
Cyclopoida		*Mesocyclops papuensis*	**X**	1	1	1	1	1	0
		Mesocyclops brooksi	**X**	0	0	0	0	1	0
		Microcyclops varicans	**O**	0	0	0	0	0	1
Calanoida	**Centropagidae**	*Calamoecia tasmanica s.l.*	**X**	0	0	1	0	0	0
Cladocera									
	Macrotrichidae	*Macrothrix cf. spinosa.*	**X**	0	0	1	0	0	0
Amphipoda									
Crangonyctoidea	**?Paramelitidae**	?Paramelitidae spp.	**S**	0	0	0	0	0	2
OLIGOCHAETA		Oligochaeta spp.	**P**	1	2	3	2	4	2
HYDRACARINA		Hydracarina spp.	**U**	1	0	3	0	3	2
INSECTA									
Ephemeroptera	**Caenidae**	*Tasmanocoenis arcuata*	**X**	0	0	3	0	0	0

				Potential exposed				Reference	
			Cat	KB50	KB56	KB57	KB58	JC90	JC91
Coleoptera	**Dytiscidae**	Dytiscidae spp. (L) (dam/imm.)	**U**	0	0	1	0	0	0
	Elmidae	*Austrolimnius* sp.	**X**	0	0	0	0	2	0
	Hydraenidae	*Hydraena sp.*	**X**	3	3	0	0	3	0
		Limnebius sp.	**X**	2	0	1	0	0	0
	Hydrophilidae	Hydrophilidae spp. (L) (dam./imm)	**U**	0	0	2	0	0	0
		Berosus sp. (L)	**X**	0	0	1	0	0	0
	Scirtidae	Scirtidae spp. (L)	**X**	0	0	2	0	0	2
Diptera	**Chironomidae**								
	Tanypodinae	*Paramerina sp.*	**X**	0	0	3	0	2	2
		Procladius sp.	**X**	0	0	2	2	0	0
		Ablabesmyia hilli	**X**	0	0	1	0	0	0
		WWT13 Unknown Genus	**U**	0	0	0	0	2	0
	Chironominae	*Cladopelma curtivala*	**X**	0	0	0	2	0	0
		Cladotanytarsus sp.	**X**	0	0	0	0	1	0
		Tanytarsus sp.	**X**	2	0	2	1	0	0
		Chironomid spp. (P)	**X**	0	0	0	0	0	3
	Ceratopogonidae	Ceratopogoniinae spp.	**X**	3	1	3	1	3	3
		Ceratopogonidae spp. (P)	**X**	1	0	2	0	0	0
	Culicidae	*Anopheles* sp.	**X**	1	0	0	0	0	0
	Stratiomyidae	Stratiomyidae spp.	**X**	0	0	1	0	0	0
		TAXA RICHNESS		**10**	**4**	**18**	**6**	**11**	**8**

Table A4-2: Wet season 2009.

				Potential exposed						Reference	
			Cat	KB50	KB56	KB57	KB58	KBNW1	KBNW2	JC90	JC91
NEMATODA		Nematoda spp.	**P**	0	0	0	0	0	0	1	0
HYDRA		*Hydra* sp.	**X**	0	0	0	0	0	2	0	0
GASTROPODA	**Planorbidae**	*Gyraulus hesperus*	**X**	3	1	0	0	0	0	0	0
CRUSTACEA											

				Potential exposed						Reference	
			Cat	KB50	KB56	KB57	KB58	KBNW1	KBNW2	JC90	JC91
Ostracoda											
		sp. a [black/green camouflage pattern]	**U**	0	1	0	0	0	0	0	0
Copepoda											
Cyclopoida		*Microcyclops varicans*	**O**	1	0	0	0	1	0	1	1
Cladocera											
	Chydoridae	*Alona cf. rigidicaudis*	**X**	0	0	0	0	0	0	0	1
Amphipoda											
Crangonyctoidea	**?Paramelitidae**	?Paramelitidae spp.	**S**	0	0	2	2	0	2	0	0
OLIGOCHAETA		Oligochaeta spp.	**P**	4	2	0	3	3	0	3	2
HYDRACARINA		Hydracarina spp.	**U**	0	0	0	0	0	0	3	2
ORIBATIDA		Oribatida spp.	**U**	2	0	0	0	0	0	0	0
INSECTA											
Collembola		Collembolla spp.	**U**	1	0	1	2	0	2	0	0
Ephemeroptera	**Caenidae**	*Tasmanocoenis arcuata*	**X**	0	0	0	0	1	1	0	0
Anisoptera		Anisoptera spp. (imm)	**U**	1	0	0	0	0	1	1	0
Coleoptera	**Hydrophilidae**	Hydrophilidae spp. (L) (dam./imm)	**U**	0	0	0	1	0	0	2	2
	Scirtidae	Scirtidae spp. (L)	**X**	0	0	2	0	0	3	0	0
Diptera	**Chironomidae**										
	Tanypodinae	*Paramerina sp.*	**X**	0	0	2	2	0	2	2	1
		Procladius sp.	**X**	0	1	0	1	0	1	0	0
	Chironominae	*Cryptochironomus griseidorsum*	**X**	0	1	0	0	0	1	0	1
		Polypedilum watsoni	**X**	0	0	0	0	0	2	1	0
		Dicrotendipes sp2	**X**	0	0	0	0	0	0	0	2
		Tanytarsus sp.	**X**	1	2	3	2	0	1	0	1
		Paratanytarsus sp.	**X**	0	2	0	0	0	0	0	0
		Unknown species WW08	**U**	0	0	0	0	0	2	0	0

			Potential exposed						Reference	
		Cat	KB50	KB56	KB57	KB58	KBNW1	KBNW2	JC90	JC91
	Chironomid spp. (P)	X	0	2	0	0	0	0	1	0
Ceratopogonidae	Ceratopogoniinae spp.	X	3	3	2	2	3	2	3	2
	Dasyheilenae spp.	X	2	1	0	0	0	1	0	3
	Ceratopogonidae spp. (P)	X	0	0	0	0	0	0	2	1
Tipulidae	Tipulidae spp.	X	0	0	0	0	1	0	0	0
	TAXA RICHNESS		**9**	**10**	**6**	**8**	**5**	**14**	**11**	**12**

Appendix 5. Macroinvertebrate taxa collected during surveys of the WPIOP study area. Values are log_{10} abundance classes, where 1=1 specimen, 2 = 2-10 specimens, 3 = 11-100, and so on.

Table A5-1: Dry season 2008.

			Potential exposed				Reference				
			KB50	KB56	KB57	KB58	Ref59	Ref60	Ref61	JC90	JC91
MOLLUSCA											
GASTROPODA	**Ancylidae**	*Ferrissia sp.*	0	1	1	0	0	0	0	0	0
	Planorbidae	*Glyptophysa (Glyptophysa) sp.*	0	1	2	0	0	0	0	2	0
		Gyraulus hesperus	0	2	0	0	0	0	0	0	1
ANNELIDA											
OLIGOCHAETA		Oligochaeta spp.	2	3	3	3	3	3	2	4	4
ARTHROPODA											
ARACHNIDA											
ACARINA		Hydracarina spp.	4	3	3	3	3	3	3	4	4
ORIBATIDA		Oribatida spp.	0	0	0	0	3	0	3	0	0
INSECTA											
Ephemeroptera		damaged spp.	0	0	0	0	0	2	0	0	0
Ephemeroptera	**Caenidae**	Caenidae spp. (imm.)	1	0	0	0	0	0	0	0	0
		Tasmanocoenis arcuata	4	3	4	2	3	2	3	3	3
	Baetidae	Baetidae spp. (imm.)	0	3	1	0	2	0	3	4	3
		Cloeon sp.	3	3	3	1	0	0	2	3	3
Odonata											
Zygoptera	**Coenogrionidae**	*Argiocnemis rubescens*	0	2	2	0	0	0	0	0	0
		Pseudagrion microcephalum	0	0	*2*	0	0	0	0	0	0
Anisoptera	**Gomphidae**	*Austrogomphus gordoni*	2	0	0	0	0	0	0	2	2
	Libellulidae	*Diplacodes haematodes*	0	2	2	0	2	1	2	0	0
		Orthetrum caledonicum	0	1	2	0	0	0	0	0	0
Hemiptera	**Belostomatidae**	*Diplonychus sp.*	0	2	0	0	0	0	0	0	0
	Corixidae	Corixidae spp. (imm.)	2	1	2	0	0	0	0	2	0
		Micronectidae sp. A	2	2	2	0	0	0	0	0	0
	Gerridae	*Limnogonus fossarum gilguy*	0	0	1	0	1	0	0	0	0
	Nepidae	*Laccotrephes tristis*	0	0	0	0	0	1	0	0	0
	Nepidae	*Ranatra occidentalis*	0	3	1	0	0	0	0	1	0

			Potential exposed				Reference				
			KB50	**KB56**	**KB57**	**KB58**	**Ref59**	**Ref60**	**Ref61**	**JC90**	**JC91**
	Notonectidae	*Anisops* sp. (F)	0	1	3	0	0	0	0	0	0
	Pleidae	*Paraplea brunni*	0	4	3	2	0	1	0	0	0
	Velidae	Microvelia sp. (F)	0	0	0	1	0	0	0	0	0
		Microvelia spp. (imm.)	1	0	0	0	1	1	0	0	0
Coleoptera	**Dytiscidae**	*Copelatus bakewelli*	0	0	1	0	0	0	0	0	0
		Copelatus nigrilineatus	0	1	0	0	0	0	0	0	0
		Cybister tripunctatus	2	2	2	0	1	2	2	2	1
		Cybister godeffroyi	0	0	0	2	2	0	0	1	0
		Eretes australis	0	1	0	0	0	0	0	0	0
		Hydroglyphus orthogrammus	0	0	1	0	0	1	0	0	1
		Hydroglyphus trilineatus	0	3	0	0	0	0	0	0	1
		Hyphydrus lyratus	1	3	2	0	0	0	2	0	0
		Laccophilus sharpi	0	2	2	0	0	0	0	0	0
		Limbodessus compactus	2	0	0	0	0	0	0	0	2
		Necterosoma regulare	2	2	2	0	0	0	0	0	1
		Tiporus tambreyi	4	3	0	3	0	0	2	3	3
		Tiporus lachlani	2	2	3	0	0	0	0	0	0
	Elmidae	*Austrolimnius* sp.	2	2	0	0	0	0	0	0	0
	Gyrinidae	*Dineutus australis*	0	0	3	0	0	0	0	0	0
	Hydraenidae	Limnebius sp.	1	2	0	1	0	0	0	2	2
		Hydraena sp.	0	0	0	0	2	3	0	2	3
	Hydrophilidae	Agraphydrus coomani	0	0	0	0	0	0	0	0	2
		Coelostoma sp.	0	0	0	0	0	0	0	0	1
		Hydrophilidae spp. (damaged/imm)	0	0	0	0	0	0	0	0	2
		Berosus dallasae	0	2	3	1	0	0	0	0	0
		Enochrus maculiceps	0	0	0	0	0	0	0	0	1
		Helochares sp. (L)	0	0	1	0	1	0	0	1	0
		Helochares tatei	0	0	0	0	0	1	0	1	0
		Laccobius billi gentili	0	0	0	0	0	0	0	1	2
		Paracymus spenceri	0	0	0	0	0	1	0	0	0
		Regimbartia attenuata	0	1	0	0	0	0	0	0	0
	Hydrochidae	*Hydrochus* sp.	1	0	1	0	0	2	0	1	3
	Limnichidae	Limnichidae spp.	0	0	0	0	0	0	0	0	2
	Noteridae	*Notomicrus tenellus*	0	0	0	0	1	0	0	0	0
	Scirtidae	Scirtidae spp. (L)	0	0	0	0	0	0	1	0	2
Diptera	**Chironomidae**										

			Potential exposed				Reference				
			KB50	**KB56**	**KB57**	**KB58**	**Ref59**	**Ref60**	**Ref61**	**JC90**	**JC91**
		Chironomidae spp. (P)	3	2	2	0	2	2	2	2	2
	Tanypodinae	*Paramerina sp.*	0	2	2	0	2	0	2	2	3
		Larsia ?albiceps	0	0	3	0	0	0	2	0	0
		Procladius sp.	0	2	3	3	3	0	2	3	2
		Ablabesmyia hilli	0	1	3	0	2	2	3	2	3
		unknown genus	0	0	0	0	2	0	2	3	3
	Orthocladinae	*Rheocricotopus sp.*	0	0	0	0	2	0	0	0	0
		Cricotopus albitarsis	1	0	0	0	0	0	0	0	0
		Thienemanniella sp.	0	0	0	0	3	0	2	0	0
		Nanocladius sp.	0	0	0	0	2	0	0	0	3
	Chironominae	*Chironomus sp.*	0	0	0	0	0	0	0	2	3
		Cryptochironomus griseidorsum	0	0	2	0	1	0	0	0	0
		Polypedilum nubifer	0	0	2	0	3	0	1	2	0
		Polypedilum (Pentapedilum) leei	0	0	0	0	1	0	2	0	3
		Dicrotendipes sp1	0	1	0	0	0	3	2	2	0
		Dicrotendipes sp2	0	0	0	0	3	0	3	0	0
		Cladopelma curtivala	0	2	2	1	3	0	2	3	3
		Polypedilum watsoni	0	0	0	0	0	0	0	1	0
		Polypedilum sp.	0	0	0	0	2	0	0	0	0
		Dicrotendipes sp.	0	0	0	0	1	0	0	0	0
		unknown genus	0	2	0	0	0	1	0	0	0
		Tanytarsus sp.	3	2	3	2	3	2	3	2	2
		Paratanytarsus sp.	0	4	3	0	1	0	0	3	0
		Cladotanytarsus sp.	4	3	3	0	0	1	0	3	2
		unknown genus	3	0	0	0	1	2	0	3	3
	Ceratopogonidae	Ceratopogoniinae spp.	0	2	3	2	2	2	3	2	2
		Dasyheilenae spp.	3	0	0	0	2	0	0	0	0
		Ceratopogonidae spp. (P)	3	0	0	0	1	0	2	3	3
	Culicidae	Culicidae spp. (damaged/imm.)	0	2	0	0	1	0	0	0	0
		Anopheles sp.	2	1	1	0	3	1	2	0	0
Trichoptera	**Ecnomidae**	*Ecnomus* sp.	0	0	2	0	3	0	1	3	1
	Hydroptilidae	*Hellyethira* spp. imm	0	0	0	0	0	1	0	0	2
		Orthotrichia sp.	0	0	0	0	0	0	2	0	0
	Leptoceridae	*Oecetis* sp.	2	1	2	0	0	0	0	0	0
		TAXA RICHNESS	**27**	**44**	**43**	**14**	**37**	**24**	**29**	**35**	**39**

Table A5-2: Wet season 2009.

			Potential exposed						Reference				
			KB50	**KB56**	**KB57**	**KB58**	**KBNW1**	**KBNW2**	**Ref59**	**Ref60**	**Ref61**	**JC90**	**JC91**
NEMATODA		Nematoda spp.	0	0	0	0	0	0	0	0	0	0	2
CNIDARIA													
HYDRAZOA		*Hydra* sp.	3	3	3	1	1	4	2	0	0	0	2
MOLLUSCA													
GASTROPODA	**Ancylidae**	*Ferrissia sp.*	0	1	0	0	0	1	0	0	0	0	0
	Lymnaeidae	*Austropeplea lessoni*	0	2	0	2	2	2	0	0	0	0	0
	Planorbidae	*Glyptophysa (Glyptophysa) sp.*	0	0	0	0	0	0	0	0	0	0	0
		Gyraulus hesperus	3	2	2	0	0	0	0	0	2	0	0
ANNELIDA													
OLIGOCHAETA		Oligochaeta spp.	3	2	0	2	0	0	0	3	2	0	3
ARTHROPODA													
ARACHNIDA													
ACARINA		Hydracarina spp.	3	0	0	3	0	2	3	4	3	4	2
ORIBATIDA		Oribatida spp.	0	0	1	0	0	0	4	0	0	2	0
INSECTA													
Ephemeroptera		Ephemeroptera spp. (dam/imm)	2	0	0	0	2	2	0	0	3	3	0
	Caenidae	Caenidae spp. (imm.)	0	0	0	0	3	0	0	0	0	0	0
		Tasmanocoenis arcuata	2	3	0	2	3	3	3	4	4	3	3
	Baetidae	Baetidae spp. (imm.)	2	0	0	3	3	0	3	0	0	0	0
		Cloeon sp.											
Odonata													
Zygoptera		Zygoptera spp. (imm)	0	0	0	0	1	1	0	0	0	0	0
	Coenogrionidae	*Argiocnemis rubescens*	0	2	0	2	2	2	0	0	0	0	0
		Pseudagrion microcephalum	0	0	0	0	0	0	0	0	0	0	0
Anisoptera		Anisoptera spp. (imm)	0	0	0	0	0	1	0	1	0	0	0
	Aeshnidae	*Hemianax papuensis*	0	0	0	0	1	3	0	0	0	0	0
	Gomphidae	*Austrogomphus gordoni*	0	2	2	0	0	0	1	0	2	2	2
	Libellulidae	*Diplacodes haematodes*	1	3	2	2	2	3	3	2	2	0	2
		Orthetrum caledonicum	2	0	0	1	2	0	0	1	1	0	1
		Nannophlebia injibandi	0	0	0	1	0	0	0	0	0	0	0

			Potential exposed						Reference				
			KB50	KB56	KB57	KB58	KBNW1	KBNW2	Ref59	Ref60	Ref61	JC90	JC91
Hemiptera	**Belostomatidae**	*Diplonychus sp.*	0	0	0	0	0	0	0	0	0	0	0
	Corixidae	Corixidae spp. (imm.)	2	2	0	2	0	2	3	3	2	0	2
		Micronecta sp. A	3	2	0	2	0	2	2	3	2	3	2
	Gerridae	Gerridae spp. (imm)	2	0	0	0	0	1	0	0	0	2	0
		Limnogonus fossarum gilguy	0	1	2	2	0	2	3	0	0	2	1
	Hebridae	*Merragata hackeri*	0	0	0	0	0	0	1	0	0	0	1
	Naucoridae	*Naucoris* sp.	1	0	0	0	0	0	0	0	0	0	0
	Nepidae	*Laccotrephes tristis*	0	0	0	0	0	0	0	0	0	0	0
		Ranatra occidentalis	1	0	0	0	0	0	0	0	0	0	0
	Mesovelidae	*Mesovelia vittigera*	0	0	0	0	0	0	2	0	0	0	0
	Notonectidae	*Anisops* sp. (F)	0	0	0	0	0	0	0	0	0	0	0
	Pleidae	*Paraplea brunni*	2	1	0	2	1	2	0	0	0	0	0
	Velidae	Microvelia sp. (F)	0	0	0	0	0	0	0	0	0	0	0
		Microvelia spp. (imm.)	0	0	0	0	0	1	2	0	0	0	0
Coleoptera		Unknown Coleoptera (L)	0	0	0	1	0	0	0	0	0	0	0
	Dytiscidae	*Allodessus bistrigatus*	0	0	0	2	1	0	0	0	0	0	0
		Copelatus bakewelli	0	0	2	0	0	0	0	0	0	0	1
		Copelatus nigrilineatus	0	0	0	0	0	0	0	0	0	0	0
		Cybister tripunctatus	0	0	0	0	0	0	1	0	0	0	0
		Cybister godeffroyi	1	0	1	0	2	0	0	2	1	2	0
		Eretes australis	0	0	0	0	0	0	0	0	0	0	0
		Hydroglyphus daemeli	1	1	2	2	2	0	0	0	1	0	0
		Hydroglyphus orthogrammus	0	0	2	1	0	0	0	0	0	0	0
		Hydroglyphus trilineatus	2	2	3	3	3	1	0	1	2	0	2
		Hyphydrus elegans	3	1	0	2	0	2	0	0	0	0	0
		Hyphydrus lyratus	1	0	1	2	3	0	0	0	0	0	0
		Laccophilus sharpi	0	0	0	2	0	0	0	0	1	0	0
		Limbodessus compactus	0	0	2	0	1	0	1	2	0	0	1
		Necterosoma regulare	0	0	0	0	0	0	0	0	0	0	0
		Tiporus tambreyi	1	0	1	0	2	0	1	0	0	0	0
		Tiporus lachlani	0	0	1	0	1	0	0	0	0	3	0
		Tiporus sp.	0	0	0	0	0	0	1	0	0	0	0
		Tiporus sp. (L)	0	0	0	0	0	0	1	0	0	0	0
		Tribe Biddessini (L)	0	0	0	0	0	0	1	0	0	0	0
	Elmidae	*Austrolimnius* sp. (L)	0	0	0	0	0	0	0	0	0	0	0
	Gyrinidae	*Dineutus australis*	0	0	0	0	1	0	0	0	0	0	0
	Haliplidae	*Haliplus pilbarensis*	0	0	0	1	0	0	0	0	0	0	0

			Potential exposed						Reference				
			KB50	KB56	KB57	KB58	KBNW1	KBNW2	Ref59	Ref60	Ref61	JC90	JC91
	Hydraenidae	*Limnebius* sp.	0	0	0	1	1	0	0	1	0	0	2
		Hydraena sp.	0	0	0	0	0	3	3	3	3	0	3
	Hydrophilidae	Hydrophilidae spp. (damaged/imm)	0	2	0	0	2	0	0	0	1	0	0
		Agraphydrus coomani	0	0	0	0	0	0	0	0	0	0	2
		Coelostoma sp.	0	0	0	0	0	0	0	0	0	0	0
		Berosus dallasae	1	0	0	2	2	0	0	0	0	0	0
		Berosus pulchellus	0	0	0	0	2	0	0	0	0	0	0
		Berosus sp (L)	0	2	0	0	1	0	0	0	0	0	2
		Enochrus deserticola	0	0	0	0	0	0	0	2	1	0	2
		Enochrus maculiceps	0	0	0	0	0	0	0	0	0	0	0
		Helochares sp. (L)	1	0	0	2	0	2	0	0	1	0	2
		Helochares tatei	0	0	0	0	0	0	0	2	3	0	3
		Laccobius billi gentili	0	1	0	1	0	0	0	0	0	0	2
		Paracymus spenceri	0	2	0	2	0	0	2	0	0	0	2
		Paranacaena sp.	0	0	0	0	0	1	0	0	0	0	0
		Regimbartia attenuata	0	0	0	0	1	0	0	0	0	0	0
		Sternolophus marginicollis	0	0	0	0	0	1	0	0	0	0	0
	Hydrochidae	*Hydrochus* sp.	1	2	0	2	2	2	2	2	2	2	1
	Limnichidae	Limnichidae spp.	0	0	0	0	0	0	0	0	0	0	0
	Noteridae	*Notomicrus tenellus*	0	0	0	0	0	0	2	0	2	0	0
	Scirtidae	Scirtidae spp. (L)	0	0	0	0	0	0	0	0	0	0	0
Diptera	**Chironomidae**												
		Chironomidae spp. (P)	2	2	2	2	2	0	0	3	2	2	2
	Tanypodinae	*Paramerina sp.*	0	2	0	2	0	0	1	2	2	1	1
		Larsia ?albiceps	1	2	2	3	3	2	3	2	3	1	0
		Procladius sp.	3	3	3	2	2	2	1	4	3	3	3
		Ablabesmyia hilli	0	2	1	2	0	2	3	0	3	0	0
		unknown genus	0	0	0	0	0	0	0	0	0	0	0
	Orthocladinae	*Rheocricotopus sp.*	0	0	0	0	0	0	0	0	0	0	0
		Cricotopus albitarsis	0	0	0	0	0	0	0	0	0	0	0
		Thienemanniella sp.	0	0	0	2	0	0	0	0	0	0	0
		Nanocladius sp.	0	0	0	0	0	0	0	0	0	0	0
		Unknown genus (WW08)	0	0	0	0	0	0	0	0	0	1	1
		Parakiefferiella sp.	0	0	0	0	0	0	1	0	0	0	0
	Chironominae	*Chironomus sp.*	3	2	1	2	1	0	0	0	0	0	2
		Cryptochironomus griseidorsum	1	2	2	1	0	1	0	0	0	0	2
		Stenochironomus watsoni	0	0	0	0	0	1	0	0	1	0	0

			Potential exposed						Reference				
			KB50	KB56	KB57	KB58	KBNW1	KBNW2	Ref59	Ref60	Ref61	JC90	JC91
		Polypedilum nubifer	0	0	0	0	0	0	0	0	3	0	0
		Polypedilum (Pentapedilum) leei	0	0	0	0	1	0	0	2	2	0	0
		Dicrotendipes sp1	3	0	0	0	0	1	2	2	1	3	3
		Dicrotendipes sp2	3	0	0	2	1	0	2	3	0	2	3
		Cladopelma curtivala	0	2	2	2	1	0	0	2	2	0	2
		?Paratendipes sp.	0	0	0	0	1	0	0	0	0	0	0
		Polypedilum watsoni	0	0	0	0	0	0	0	0	0	0	0
		Polypedilum sp.	0	1	0	0	0	1	0	0	2	0	0
		*Parachironomus*sp. (?K2)	0	0	0	0	0	0	1	0	0	0	0
		Dicrotendipes sp.	0	0	0	0	0	0	0	0	0	0	0
		Kiefferulus intertinctus	0	0	0	0	0	0	0	0	2	0	0
		unknown genus	0	0	0	0	0	0	0	0	0	0	0
		Tanytarsus sp.	3	3	4	3	3	3	2	3	2	4	3
		Paratanytarsus sp.	3	2	2	3	0	2	2	0	0	0	0
		Cladotanytarsus sp.	2	0	0	0	3	1	2	0	0	0	0
		unknown genus	0	0	0	0	0	0	0	0	1	0	0
	Ceratopogonidae	Ceratopogoniinae spp.	3	3	2	3	3	3	2	2	2	0	3
		Dasyheilenae spp.	0	1	2	3	0	1	3	0	0	0	0
		Ceratopogonidae spp. (P)	0	0	0	0	0	0	1	0	0	0	0
	Culicidae	Culicidae spp. (dam./imm.)	0	0	0	0	0	0	0	0	0	0	0
		Anopheles sp.	0	0	0	2	0	2	0	0	1	0	0
		Culex sp.	0	0	0	0	1	0	0	0	0	0	0
	Ephydridae	Ephydridae spp.	0	0	1	0	0	0	0	0	0	0	0
	Stratiomyidae	Stratiomyidae spp.	0	0	0	0	0	0	1	0	0	0	0
	Tabanidae	Tabanidae spp.	0	0	0	0	0	1	0	0	0	0	0
	Tanyderidae	Tanyderidae spp.	0	0	0	0	0	0	0	0	0	0	1
Trichoptera	**Ecnomidae**	*Ecnomus* sp.	1	0	0	1	0	0	0	0	2	0	1
	Hydropsychidae	*Cheumatopsyche sp. AVII*	0	0	0	2	0	0	0	0	0	0	0
	Hydroptilidae	*Hellyethira* sp.	3	0	0	2	0	0	0	0	0	2	3
		Orthotrichia sp.	0	0	0	0	0	0	2	1	2	0	0
	Leptoceridae	*Oecetis* sp.	0	0	0	0	0	0	0	0	1	0	1
Lepidoptera	**Pyralidae**	Nymphulinae spp.	0	0	0	1	0	0	1	0	0	0	0
		TAXA RICHNESS	**37**	**34**	**27**	**48**	**40**	**38**	**40**	**27**	**40**	**20**	**40**