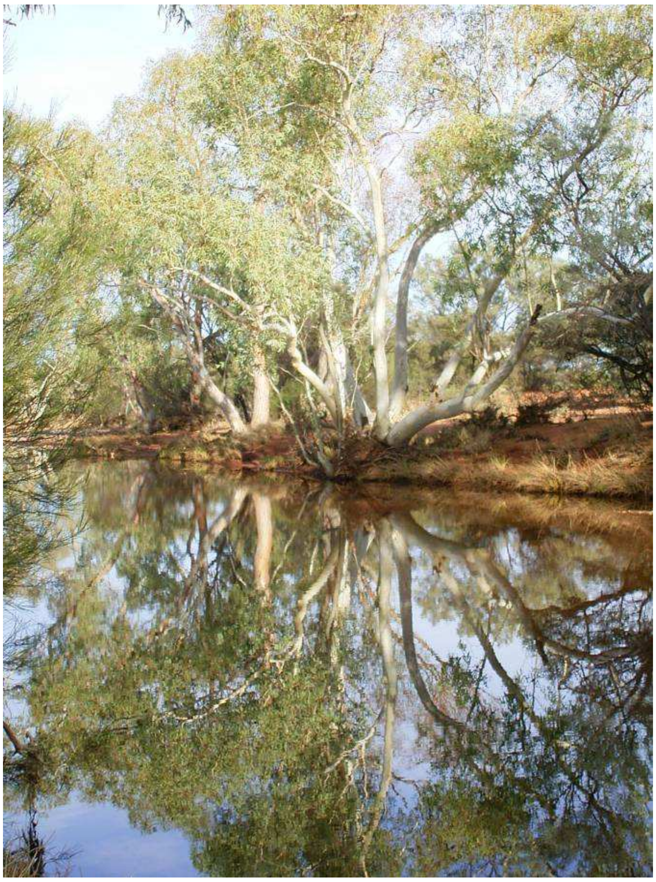
Yakabindie Nickel Project

Baseline Aquatic Biology and Water Quality Study of Jones Creek, including the South-west Claypan Area



prepared for

bhpbilliton

by

Wetland Research & Management

Baseline Aquatic Biology and Water Quality Study of Jones Creek, including the South-west Claypan Area

Prepared for:

Nickel West, a division of BHP Billiton

191 Great Eastern Highway, Belmont, WA 6104
Ph: (61 8) 9479 0500, Fax: (61 8) 9479 0641

By:

Wetland Research & Management

28 William Street, Glen Forrest, WA 6071, Australia
Ph (61 8) 9298 9807, Fax (61 8) 9380 1029, e-mail: awstorey@cyllene.uwa.edu.au

In Association with:

Sinclair Knight Merz

9th Floor Durack Centre, 263 Adelaide Terrace, PO Box H615, Perth WA 6001, Australia
Ph (61 8) 9268 4400, Fax (61 8) 9268 4488

Final Report

16/09/05

Frontispiece: (clockwise) main picture, *Eucalyptus camaldulensis* at confluence of northern and eastern tributaries; adult burrowing frog *Cyclorana platycephala*; main channel of Jones Creek at Goldfields Highway crossing (Site 2); shield shrimp *Triops australiensis australiensis*.

Study Team

Management: Andrew W. Storey
Field Work: Andrew W. Storey & Sue Creagh
Macroinvertebrate Identification: Jess Lynas
Micro-invertebrate Identification: Russ J. Shiel (University of South Australia)
Data analysis: Sue Creagh & Jess Lynas
Report: Sue Creagh & Andrew W. Storey

Acknowledgements

This project was undertaken by *Wetland Research & Management* (WRM) on behalf of Sinclair Knight Merz for Nickel West, a division of BHP Billiton. WRM would like to acknowledge the following UWA researchers for assistance with taxonomy; Dr Don Edward (Chironomidae), Dr Brenton Knott (Anostraca) and Martin Dziminski (frogs). The maps of the study area were provided by Sinclair Knight Merz. Jenny Yan (SKM) is thanked for constructive criticism on the draft report and for her efficient overall management of this project on behalf of SKM. We are grateful to Doug and Lucy Brownlie (managers of Yakabindie Pastoral Lease) for access to wetlands within property boundaries. John Tomich, Kevin Shugg, Kim Brand and Leisa Turner (all Nickel West, Mt Keith) are thanked for kindly providing assistance on-site. Finally, our thanks to Keith Berry (Nickel West, Perth) who provided data and text on the hydrology and hydrogeology of Jones Creek.

Recommened Reference Format

WRM (2005). Yakabindie Nickel Project: Baseline Aquatic Biology and Water Quality Study of Jones Creek, including the South-west Claypan. Unpublished report by Wetland Research & Management to BHP Billiton. June 2005.

Disclaimer

CONTENTS

List of Figures, Tables and Plates

Figures

Tables

Plates

EXECUTIVE SUMMARY

Nickel West propose to mine nickel at Yakabindie, 40 km north of Leinster in the north eastern Goldfields of Western Australia. Under the proposal, the main channel of the ephemeral Jones Creek will be diverted to accommodate the main open cut pit and substantial waste rock dumps will be constructed in the catchment. The development has the potential to adversely affect the aquatic ecosystem of both the creek and the downstream claypans into which the creek ultimately flows. The goal for Nickel West is to maintain the natural biodiversity of these temporary waters. To that end, the aquatic fauna and physico-chemistry of eight sites along Jones Creek and one in the south-west claypan area were sampled in May 2005 by *Wetland Research & Management* (WRM). Sampling represented a baseline study to help establish conditions prior to commencement of mining, against which future changes (if any) may be assessed.

The specific aims of the project were to identify the resident fauna of the creek and claypan to the lowest possible taxonomic level, compare biodiversity and invertebrate assemblages amongst sites and relate differences in fauna to differences in water quality. Tadpoles and aquatic macro- ('benthic') and micro- ('planktonic') invertebrates were collected using qualitative sweeps. Sampling for fish was conducted using baited box traps and by direct observation. Broad, qualitative assessments of riparian habitat condition were made on the basis of dominant plant species and erosional characteristics.

Riparian Habitat Condition

Jones Creek catchment was considered extremely degraded due to historic pastoral practices and unrestricted livestock access to the natural waterbodies. Channels at all sites were characterised by poor bank stability with extensive erosion - bank slumping, channel widening and bed down-cutting. Understorey vegetation was, at best, sparse with much exposed soil. Along the creekline, tree species comprised a sparse to open woodland of *Eucalyptus camaldulensis* with the Priority One *E. striaticalyx* subsp. *delicata* (the ID & priority status is being checked) present in the northern headwater sub-catchment. Most trees observed were mature, with little evidence of recruitment. The regional ecological value of the remnant riparian vegetation was considered to be high, in particular the eucalypts, as they represent the only large tree species within the catchment. The remnant vegetation is likely to provide at least some of the energy ('food source') that drives many aquatic processes (*e.g.* food webs) as well as providing food, shade and shelter for both terrestrial and aquatic fauna. The trees and perennial shrubs, such as mulga (*Acacia aneura*), also help stabilise creek banks against further erosion.

Physico-chemistry

Values for most of the physico-chemical variables measured were within the ranges expected for 'slightly-moderately disturbed' ecosystems (ANZECC/ARMCANZ 2000a) with no significant differences between control and exposed sites. However, multivariate analysis showed the physico-chemistry of the claypan to be clearly different to that of Jones Creek.

Waters in Jones Creek were clear, while those of the claypan were very turbid (Secchi 6.0 cm; turbid. 320 NTU). All waters were fresh, with conductivities of 52 - 132 µS/cm, water temperature of 18.5 – 22.3 °C, dissolved oxygen levels of 74% to 144% (super-saturated) and pH ranging from 5.1 - 6.8, with lowest pH occurring at Site 5 at the old highway crossing. The composition of major ions was noticeably different in the claypan compared with Jones Creek sites; the claypan was dominated by potassium and bicarbonate ($K^+ > Na^+ > Ca^{2+} > Mg^{2+}$: $HCO3^- > Cl^- > SO_4^{2-}$) whereas the creek sites were dominated by sodium/calcium and bicarbonate ($Na^+ <> Ca^{2+} > K^+ \geq Mg^{2+}$: $HCO_3^- > Cl^- > SO_4^{2-}$).

Total nitrogen concentrations ranged from 0.46 - 2.4 mg TN/L, nitrates ranged from 0.06 – 1.9 mg NO_3/L and total phosphorus 0.02 to 0.06 mg TP/L. Soluble reactive phosphorus (SRP) was at or below the detection level of 0.01 mg/L for filterable reactive phosphorus (FRP). TN, NO_3 and TP concentrations exceeded ANZECC/ARMCANZ (2000a) guideline values at all sites. Nitrogen levels were particularly high at Site 6, west of Goldfields Highway. Maximum TP concentrations were recorded from the uppermost control site (Site 1), along the northern tributary.

Most metal concentrations in the water column were below ANZECC/ARMCANZ (2000a) guidelines with the exception of Cr, Cu, Ni and Zn. All concentrations measured during the current study should be treated as background levels or 'reference' levels against which to compare any future monitoring results. The relatively high levels of Cr, Cu and Ni in claypan waters and/or sediments may indicate a tendency for these metals to accumulate here. Nickel West has already completed a baseline study to determine pre-mining sediment metal chemistry in both Jones Creek and the claypan (SKM 2005). It is anticipated that sediment metal chemistry will be monitored in the future. The more acidic waters (pH 5.1) at Site 5 (main channel at the old highway) may indicate a greater potential for mobilization of some metals from bed sediments at this site.

Aquatic Fauna

In total, 124 taxa were recorded from the eight sites sampled in May 2005, representing a highly diverse fauna for inland waters. The fauna was dominated by insects (37%), rotifers (33%) and crustacea (17%). Chironomidae (non-biting midges) constituted 28% of all insect taxa. Of species recorded, 63% were considered to be 'permanent' residents with desiccation-resistant life stages that would allow them to remain within the creek or claypan once surface waters had evaporated. The majority of these species were the micro-crustacea (protists, rotifers, copepods & ostracods) and branchiopods (notostracans, anostracans, conchostracans & cladocerans). Species diversity at individual sites ranged from 33 in the claypan to 62 in Jones Creek.

Multivariate analyses (PRIMER) indicated the invertebrate community structure of the claypan to be distinct from Jones Creek. Analyses of Jones Creek showed spatial variability of 'permanent' aquatic invertebrate fauna was low in comparison with 'temporary' invertebrate fauna. There were no significant differences in assemblages or taxa richness of permanent fauna, with high between-site similarity in assemblage composition for control and exposed sites. The inter-site variability in assemblage composition were possibly influenced by local variations/between-pool differences in turbidity, ionic composition, nutrient levels and concentrations of Cr, Zn and Ni, as indicated by PRIMER analysis.

Most invertebrate species recorded were considered cosmopolitan and widespread throughout inland waters in arid regions of Western Australia. One new species of rotifer (*Cephalodella* sp. nov.) and range extensions (new to WA) for three other rotifers were recorded, however, there have been few comprehensive studies of the taxonomy and distribution of microfauna in Western Australia, making it difficult to confer conservation status on these species. The apparent restriction of five rotifer and one cladoceran species was considered an artefact of the sampling and they are likely to be present at all creek sites, but by chance were only recorded in the sub-samples from Site 3 (main channel near Six Mile Well).

No fish species were recorded, but tadpoles of the burrowing frog genus, *Neobatrachus* sp. (possibly *N. wilsmoorei*, *N. suter* and/or *N. sudelli/centralis*), were present at all sites and were particularly abundant in the main channel of Jones Creek. Two adult water holding frogs, *Cyclorana platycephala*) were also recorded from Jones Creek. All species are common in arid zones across central Western Australia.

Recommendations

A number of recommendations are made in regard to the proposed diversion channel, with particular reference to design features in order that the channel approximates a more 'natural' creekline. Minor modifications to the sampling programme are also recommended, including the targeting of permanent fauna (*e.g.* micro-invertebrates and branchiopods) for long-term bio-monitoring. This group is more likely to show effects of any environmental change than are temporary residents which can avoid unfavourable conditions.

1. INTRODUCTION

1.1 Study Area

As part of a proposal to establish a nickel mine at Yakabindie, Sinclair Knight Merz (SKM), on behalf of Nickel West commissioned *Wetland Research and Management* (WRM) to conduct baseline biological surveys of temporary waterbodies potentially affected by the mine development. The proposed development is located approximately 50 km north of Leinster, on a pastoral lease east of the Goldfields Highway and immediately west of the Wanjarri Nature Reserve. The current study will contribute baseline data prior to commencement of open pit mining of two nickel deposits, namely Six Mile and Goliath. Mining of the Six Mile deposit will require the diversion of headwater channels of Jones Creek. Nickel West envisage that operations will commence at Yakabindie in 2007.

The mine lease lies within the rangelands of the north-eastern Goldfields region of Western Australia. The climate is arid Mediterranean with hot summers, cool winters, unpredictable rainfall and high evaporation rates (2,400 mm/year). Most rainfall is typically associated with south-westerly frontal activity during winter (May – August), however tropical cyclones to the north, result in rain bearing depressions which bring high summer falls of more than 70 mm/day.

The study area forms part of the Western Plateau drainage division, which covers more than half of the State and more than a third of the continent, yet has virtually no large river systems (Rangelands NRM Co-ordinating Group 2005b). The major drainage line within the study area is the highly ephemeral Jones Creek. Jones Creek flows southwest through the pastoral lease along a low-lying, broad alluvial valley. The eastern headwaters drain the Wanjarri Nature Reserve. The creek is endorheic[1] and west of the Goldfields Highway the channel becomes increasingly diffuse and ill-defined, flowing into small claypans and ultimately the larger playa of Lake Miranda, 17 km south-west of the proposed mine.

1.1.1 Hydrogeology

Jones Creek and Lake Miranda lie within the catchment of a large, ancient river system, the Carey Palaeodrainage, which once flowed south-easterly into the Eucla Basin that now lies beneath the Nullabor Plain (Johnson *et al.* 1999). The palaeodrainage contains major aquifers with groundwaters ranging from fresh to hypersaline. Groundwater resources within the region include fractured rock, calcrete and unconfined alluvial aquifers, all of which have been described in detail by Johnson *et al.* 1999, and at least some of which are likely to be important in maintaining local stygofauna and phreatophytic[2] vegetation. A review of groundwater research in the region is also given in Rangeland NRM Co-ordinating Group (2005b).

The upper part of the Jones Creek catchment where mining activities will occur is an area of outcropping greenstone bedrock. Alluvial cover is less than 3 m in thickness, mainly forming narrow and shallow sinuous deposits associated with the creek beds. Groundwater occurrence within the basement is in narrow, but widely spaced fracture zones and porous secondary regolith materials formed as caprock on the pods of dunite rock type, which host the nickel deposits. These water bodies are minor, hydraulically isolated aquifers and do not form a regionally continuous aquifer system, nor are they of particular significance to the groundwater resources of the region. Depth to water in these minor aquifers is 10's of metres below ground level. In the locality of the Six Mile deposit, the ground water level is about 404 m AHD or 15 - 30 m below ground level.

[1] Rivers or creeks that do not reach the sea, but flow 'inland' to terminal wetlands or claypans.
[2] Groundwater dependent.

As a result of preceding rainfall, creek bed alluvial deposits become fully saturated with water resulting in the creek flowing. As these small accumulations are subsequently gradually depleted by evaporation, transpiration by vegetation and by downward leakage to the minor bedrock aquifers, the creek ceases to flow. Remaining pools which are surface expressions of the saturated alluvium gradually disappear and the creek dries, leaving only subsurface moisture in the alluvial deposits. These 'perched' water accumulations are controlled by rainfall and runoff and it is unlikely that they would be affected by any mine dewatering drawdown in the bedrock aquifers.

1.1.2 Hydrology

Jones Creek is ephemeral, flowing briefly in response to unusually intense or prolonged rainfall. Nickel West has installed monitoring systems to develop the statistical relations between rainfall and runoff. In general, creek flow may occur in response to rainfall of about 25 mm or more if the intensity is sufficient. Rainfall totals of 52 mm in a month, 43 mm over 10 days or 35 mm over 3 days each have a return period of 1 year, *i.e.* are likely to occur in most years and would typically result in small flow events.

For peak flow, the critical duration of rainfall causing runoff is 2 - 3 hours. In early May 2005, a 1 in 5 year rainfall event of 2 -3 hour duration generated flows of 108 m^3/sec and flow velocity up to 1.7 m/sec at the gauging station at the old highway crossing. Flows in response to 1 in 100 year rainfall over the critical period are likely to be 2.5 times greater, with velocity at the gauging station potentially reaching 2.2 m/sec.

The creek and floodplain claypans become interconnected only during large rainfall events. Such flooding is likely to facilitate nutrient and carbon exchange between the floodplain and the water bodies and between the hyporheic zone[3] and surface sediments (Claret & Boulton 2003). Flooding is also likely to allow passage of aquatic organisms and provide a mechanism for gene flow between water bodies that remain dry and isolated for much of the year, or even years (Jenkins *et al*. 2002).

1.1.3 Catchment Condition

Soils of the Jones Creek catchment are dominated by duplex soils, relictual, sandy red loams and gravels over clay, with naturally low nutrient and moisture content. These relatively shallow, poorer soils support only sparse to moderately dense vegetation dominated by mulga (*Acacia aneura*) woodlands and hummock grasslands (*e.g.* Wanderrie grass associations) with an open cover of river red gums (*Eucalyptus camaldulensis*) along the creekline. The Priority One[4] *Eucalyptus stratiocalyx* subsp. *delicata* (checking ID and status) is also present along the creekline in headwater regions of the northern tributary. Within this sub-catchment, *E. striaticalyx* subsp. *delicata* occurs on calcareous soils which are highly eroded and inherently unstable (Geoff Cockerton, Western Botanical, pers. comm.).

The south-west claypans are fringed by open thickets of Kurara (*Acacia tetragonophylla*) which are periodically inundated by rising flood waters from the claypans. Generally, soils supporting mulga woodlands are believed naturally stable but highly susceptible to erosion around stock and

---

[3] Zones of saturated sub-surface sediments beneath and beside the stream channel (Claret & Boulton 2003) that typically support micro-flora and fauna communities (*e.g.* fungi, bacteria, algae, microinvertebrates).

[4] Department of Conservation and Land Management (CALM) list of Priority Fauna, being species that are not considered Threatened under the WA Act but for which the Department feels there is cause for concern. Priority 1 taxa = taxa with few, poorly known populations on threatened lands.

vehicle tracks and 'washout' due to excessive overland flows following vegetation loss (Burnside *et al*. 1995).

From a waterways perspective, it is generally perceived by the broader community that the creeks of this fragile landscape are characterised by wet season floods and high sediment loads, reflecting combined effects of high intensity rainfall and sparse vegetative cover binding the soil together. However, it is a fallacy to assume that these rangeland catchments always had a high run-off coefficient, that water never remained on the land for long and that high rates of erosion are a natural component. The pre-European landscape has been extensively modified by grazing, particularly during the 1930's (Wilcox 1960). Between the Goldrush of the late 1800's – early 1900's and the Depression of the 1920's/1930's, large amounts of timber were also harvested for fuel and settlements buildings (Rangelands NRM Co-ordinating Group 2005b). Clearing and over-stocking has resulted in excessive catchment erosion and land degradation (Dept. of Ag. 1992, Hall *et al.* 1994, Burnside *et al.* 1995). Only a third of the north-eastern Goldfields region was considered to be in "good" condition when assessed in 1988 (Dept. of Ag. 1992, Rangelands NRM Co-ordinating Group 2005b). While some improvement has been observed since the 1990's, Pringle and Tinley (2003) believe landscape degradation is on-going.

The Water & Rivers Commission (WRC 1997) considered that the most serious cause of river degradation in northern parts of the State, was the removal of the natural riverine and catchment vegetation resulting in increased rates of runoff, leading to soil erosion and sedimentation of the rivers and reduced rainwater infiltration across the floodplain. It was considered that heavy grazing pressure over wide areas had destroyed the vegetation and exposed the fragile soil structure to the impact of periodically intense rainfall (WRC 1997). This had resulted in widespread sheet and gully erosion across much of the rangelands, and once started, gully erosion in particular was very difficult to halt. Erosion was then exacerbated by stock trampling. Once eroded from the landscape, soil washed into the rivers, making them highly turbid when flowing, and was then deposited along creek banks and in pools.

1.1.4 Previous Aquatic Fauna and Flora Surveys

There have been very few studies of the aquatic biota of the region and none that pre-date the expansion of the pastoral industry in the 1930's. Most relevant of recent studies, were comprehensive microflora/fauna surveys of Lake Miranda conducted by Curtin University (see John *et al*. 2000, 2003) and zooplankton and macroinvertebrate surveys of Lake Carey in 2003 (B. Timms, University of Newcastle, pers. com.). Under an earlier proposal by Dominion Mining Ltd to mine nickel at Yakabindie, Jones Creek was assessed as part of terrestrial flora/fauna surveys undertaken by *ecologia* (1990) in March 1990 and aquatic fauna surveys undertaken by Streamtec (1992) in January 1992. As part the current nickel mine proposal, a study of stream sediment characterisation and geomorphology of Jones Creek and the adjacent claypans was undertaken by SKM in December 2004 (SKM 2005). Vegetation surveys of the Jones Creek catchment have also recently been conducted by Western Botanical (division of Landcare Holdings Pty Ltd).

1.2 Study Objectives

The proposed locations of mine pits and waste rock dumps are all within the headwater sub-catchments of Jones Creek. The main pit (Six Mile Pit) will encompass reaches of the northern tributary, with the Waste Rock Area positioned adjacent and east of the pit and immediately upstream of the eastern tributary headwater region. Nickel West plan to divert the main channel to the south-east and around Six Mile Pit, with the diversion channel re-entering the natural creek downstream of the pit.

The proposed diversion will directly affect surface flows and there is the potential for indirect impacts through changes to overland sheet flows, further vegetation-loss and increased soil contaminants, dust and accelerated rates of erosion. All water bodies, even those that are episodic, are of great ecological significance in arid and semi-arid landscapes, supporting not only aquatic ecosystems, but terrestrial ones as well. In order to help effectively monitor any future adverse environmental effects in Jones Creek and the downstream claypans, specific aims of the current baseline study were to:

i). Identify the aquatic fauna in the receiving environments of Jones Creek and the south-west claypan area to the lowest possible taxonomic level;
ii). Determine conservation status of the aquatic fauna and compare biodiversity and invertebrate assemblages amongst sites;
iii). Relate differences in fauna to differences in physico-chemical parameters;
iv). Where possible, make comparisons with samples previously collected by Streamtec Pty Ltd in January 1992;
v). Design a long-term monitoring programme that would allow for future evaluation of levels of natural variation and assess the impacts, if any of mining.

2. METHODS

2.1 Bio-monitoring in Temporary Waters

Temporary, or ephemeral, waters are frequently cited as having lower diversity, but often higher abundances of fauna in comparison to permanent or even seasonal waters (Sheldon *et al.* 2002). In Australia however, temporary waters may support species richness not unlike that found in more permanent streams and with a high degree of endemism (Lake *et al.* 1985, Boulton & Suter 1986, Davis *et al.* 1993, Pontin & Shiel 1995, Williams 1998, 2002, Shiel *et al.* 2002). Life histories and/or survival strategies of aquatic species are intrinsically linked to seasonality and predictability of flow regimes (Clifford 1966, Williams & Hynes 1977a,b, Towns 1985, Boulton & Suter 1986, Boulton & Lake 1988, Bunn 1988, Boulton 1989, Bunn *et al.* 1989, Delucchi & Peckarsky 1989, Sheldon *et al.* 2002). Bayly and Williams (1973) categorized temporary streams on the basis of flow predictability:

- Intermittent streams – streams with seasonal flow typically occurring in semi-arid regions;
- Episodic streams – streams with unpredictable flow, occurring only after heavy rains.

Both types show distinctive aquatic faunal communities compared to permanently flowing streams. In many inland systems, species must survive in waters that either dry out completely or are reduced to a series of stagnant pools, while at other times flash flooding may result in extremely high water velocities that can dislodge and sweep fauna downstream. Despite such harsh conditions, many invertebrate species are found only in temporary streams (Bunn *et al.* 1986, 1989). Some species, including those that possess short maturation times, survive the dry season as terrestrial adult stages (*e.g.* mayflies, dragonflies, caddis flies and some beetles). Others have resistant spore/egg or larval phases and/or are capable of burrowing into moist sediments of the hyporheic zone, below stones or into decomposing wood debris (*e.g.* microcrustacea, shield shrimps, fairy shrimps, mayflies, chironomids, worms, and fly larvae). For example, all micro-invertebrates have drought-resistant eggs/spores – 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 drought over years or decades. Copepods have even been hatched from 330-year-old sediments (Hairston *et al.* 1995).

Many bivalves and gastropods are resistant to desiccation, sealing their shells with opercula or epiphragms (mucous plugs). The majority of native fish require permanent water, only colonising ephemeral streams from adjacent permanent waters during wet season flows. Exceptions are species that can aestivate in the bed of wetlands, but these species are rare and restricted in distribution to the south-west of the state. Arid-zone frogs however, are numerous and widespread across Australia, escaping desiccation by burrowing and aestivating in mucus cocoons or by sheltering in cool, moist rock habitats of ranges and breakaways.

The relative success of each of the above strategies and subsequent recruitment and ecological succession will vary from year to year depending on such factors as weather and predator avoidance, and large variations in community structure and composition may ensue in the absence of anthropogenic disturbance. Long-term temporal variation in aquatic communities has not been well-documented in Australia. In more permanent, less dynamic systems, temporal persistence may be high when assessed at family-level while at the same time, very low for individual species (Metzeling *et al.* 2002). It may be that communities of ephemeral streams display less long-term variation as resident species are likely pre-adapted to extremes in environmental conditions. Paucity of knowledge regarding the ecology of temporary waters in Australia means most bio-monitoring regimes have been developed based on the ecology of more permanent waters and are generally not well-suited to assessing degradation in arid and semi-arid environments (Boulton & Lake 1988, Boulton 1989). Smith *et al.* (2004) have provided a comprehensive critique of biotic and abiotic approaches used in assessing temporary waters Australia-wide, including the problems associated with the application of current ANZECC/ARMCANZ (2000) water quality guidelines[5] to intermittent and episodic systems. This critique was commissioned by the Australian Centre for Mining Environmental Research (Qld) in order to address some of the problems associated with assessing mine impacts in arid and semi-arid regions of Australia, and can be regarded as industry standard for monitoring temporary waters.

Recommendations by Smith *et al.* (2004) for monitoring approaches included tailoring existing methods to specific sites rather than attempting a blanket approach. Ephemeral streams are likely to have unique hydrological cycles and hence unique ecological responses to disturbance that will also be dependent upon habitat/refugia availability (*e.g.* sediment types, amount of leaf litter, riparian vegetation cover) (Towns 1985). Smith *et al.* (2004) viewed the concurrent monitoring of water quality and biota as advantageous as the biota integrates pulsed inputs over time that may be missed by spot sampling of water chemistry alone. Effects of pulsed input of pollutants may be cumulative in biota even though concentrations in the water column quickly return to background levels. The use of micro-invertebrates rather than reliance on macroinvertebrates as sole bio-indicators was also recommended, together with greater emphasis on the hyporheos rather than just the benthos[6] and water column. While benthic macroinvertebrates are the more conspicuous fauna and the most frequently studied, many are highly mobile and often only temporary residents, their presence dictated by stochastic processes, *i.e.* distance to nearest permanent water from where they can colonise, presence of adults in the refuge population, conditions that promote dispersal such as wind direction and speed, chance meeting of male and female adults at the new waterbody to produce viable eggs *etc.*

---

[5] The ANZECC guidelines specify biological, sediment and water quality guidelines for protecting the range of aquatic ecosystems, from freshwater to marine (ANZECC/ARMCANZ 2000). 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).

[6] Used here as pertaining to organisms dwelling on or in surface sediments *cf* sub-surface hyporheos.

Micro-invertebrates however, are known to play a significant role in the ecology of dry-land waters and because they generally have desiccation-resistant eggs/spores, they typically comprise the majority of fauna in episodic systems (Shiel *et al.* 2002). They are more likely to be permanent residents of the water body as either obligate or refuging members of the hyporheos and hence more likely to provide an early warning of contamination. In either case, the sensitivity to pollutants needs to be better researched together with an understanding of the life history strategies of both micro- and macroinvertebrate species, if their use as bio-indicators is to be effective.

Ephemeral streams have less capacity for dilution of mine contaminants. The 'boom and bust' nature of the hydrological cycles also mean contaminants may be transported many kilometres from the mine during flood flows, only to be concentrated by evaporation as flood waters recede. This is particularly relevant in systems that are actively eroding and in endorheic systems such as Jones Creek which terminates in claypans. The dry sediments and interstitial waters of creek beds and claypans in receiving environments may act as sinks for many pollutants and accumulate them over time, leading to greater potential for toxicological effects on both the hyporheos and surface water communities (Smith *et al.* 2004). The finer sediments of the claypans within the Jones Creek catchment give them a greater capacity to adsorb contaminants (SKM 2005).

Based on the review by Smith *et al.* (2004), methods deemed most appropriate to the current study at Yakabindie were considered to be:

- Standardised spot sampling for water quality in association with aquatic fauna sampling, taking into consideration the volume of standing water present (*i.e.* estimates of pool size; flow rates);
- Standardised semi-quantitative sampling for both micro- and macroinvertebrates identified to species-level;
- Assessment of micro- and macroinvertebrate species present as suitable indicators of mining impacts;
- Assessment of potential for artificial substrate and/or hyporheic core samplers for use in long-term monitoring of micro- and macroinvertebrate fauna;
- Assessment for potential use of propagule (resistant eggs/life phases) bank sampling of sediments for use in long-term dry-phase monitoring of micro- and macroinvertebrate fauna.
- Assessment of ecosystem function approaches to monitoring (*e.g.* metabolism chambers; stable isotope analysis).

Other methods that have been employed in temporary waters were considered less suitable due to elevated costs and technical difficulties associated with their implementation and/or the additional research and development needed to verify the approach. For example, the use of periphyton (including diatoms) as a bio-indicator requires specialist taxonomic expertise and if artificial substrates were to be used as platforms for colonisation and subsequent sampling, they would require *in situ* deployment and subsequent retrieval within a 10-14 day period. Use of metabolism chambers to measure ecosystem function (as opposed to ecosystem patterns such as species diversity & abundance) is technically challenging, requires greater site replication, often with highly variable results that are difficult to interpret (Smith & Storey 2001). Stable isotope analysis (nitrogen & carbon isotope ratios in fauna and flora) however, may prove a more useful and cost effective ecosystem function approach if potential contaminants of mining include known photosynthetic or respiratory inhibitors (*e.g.* copper).

Direct toxicity assessment (DTA) requires time and suitable bioassay species, is more suited to testing mine discharge waters prior to release and may not be relevant to whole communities nor reflect final toxicities in the receiving environments. Future monitoring may be supplemented using 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. In accordance with the ANZECC/ARMCANZ (2000) water quality guidelines, DGTs may be used as a speciation measurement to provide a better estimate of the bio-available metal concentration if the total and dissolved metal concentrations exceed the guideline trigger values. Use of DGTs may be of benefit in the future to assess levels of bio-available metals, if monitoring detects either impacts on fauna or elevated metals in water samples. Even if elevated total metals are recorded, most may be bound to suspended particulates or sediment and may not be bioavailable. It is possible that dissolved metals may also be quite high but remain unavailable through complexing with dissolved organic carbon (*e.g.* tannin). Thus analysis of total concentration, even total dissolved, may be misleading.

2.2 Sites and Sampling Design

The sampling design for the current project was based on two complementary approaches to bio-monitoring: (i) using a classic, hypothesis-testing framework by way of the Before-After-Control-Impact class of design (BACI), and (ii) multivariate analysis of changes in community structure. The concept behind BACI designs is outlined below. Multivariate analyses are discussed in Section 2.8 'Data Analysis'. Both approaches rely upon sampling replicate sites to characterise spatial variability in the parameters being measured (*i.e.* species richness, assemblage composition, water quality parameters), and to provide statistical power (*viz.* the ability to statistically detect differences/affects if they exist). Sampling in May 2005 intended to establish baseline conditions and provide data upon which an ongoing, monitoring programme could be developed.

To maximise the biodiversity recorded, provide replicate sites satisfying the above design requirements, and provide a geographical spread along the main braided drainage system with a range of different physical characteristics and types of remnant vegetation communities, a total of eight sites were sampled along the watercourse. Seven sites were chosen along Jones Creek (six on the main stem and one on a smaller tributary), and one in the northern-most claypan southwest of Goldfields Highway (Table 1 & Figure 1). Each site comprised a ~200 m reach of stream channel, with sampling conducted in isolated pools over this reach, as described below. Sampling was conducted between 18 - 20th May, approximately 10 - 14 days after good rains resulted in flow in the previously dry stream bed.

At the request of Sinclair Knight Merz, a depth gauge in the form of a PVC pipe attached to a star picket was positioned within the claypan site in order to gain some estimate of the rate of evaporation of waters from the claypan. Location of the depth gauge is given in Table 1.

Figure 1. Location of sampling sites in Jones Creek and the south-west claypan area.

REFER ACCOMPANYING PDF FILE: Field_site_locations.pdf

2.2.1 BACI Design

The basis of the BACI design is a comparison through time, before and after mine development, between fauna community responses from control and exposed (potential impact) sites (Table 1) using classic analysis of variance (ANOVA) and multivariate test methods. If disturbances from mining were evident in future, this would be detected by departure in biological responses between control and exposed sites compared to the same responses measured (i) from the same sites before development and (ii) in control sites throughout the monitoring period. Thus all sites sampled during May 2005 serve as controls prior to mining.

Sites upstream from the proposed mine site will act as future controls and Sites 4, 5, 6, 7 & 8 located in the downstream receiving environments of Jones Creek and the south-west claypans represent exposed sites. Site 8 was positioned to detect any point source contaminants entering the eastern tributary, whereas Site 5 would indicate the combined effect of point sources along both the northern and eastern tributaries. Currently, Sites 2 and 3 are located within the footprint of the future main pit. This was intentional, to identify any taxa restricted to this location and which would be affected by pit construction. These sites will be relocated upstream as the pit expands. Relocating sites in the Jones Creek environment is not considered a flaw in the design. The creek bed is highly mobile, coarse and fine sand over finer silt and clay alluvium. Because of the nature of the system, whereby it flows for short periods only, sampling will target residual pools after flow has ceased. Given the mobile nature of the bed, the size and position of these pools will vary seasonally and annually. Therefore, the exact position of sampling sites will vary as the bed of the creek varies. Sampling sites therefore will consist of adjacent pools over a 200 m reach of stream bed in the vicinity of the designated sampling location.

Table 1. Co-ordinates of sampling sites along Jones Creek and the southwest claypan

Location	Site No.	BACI Code		Latitude WGS84	Longitude WGS84
Northern tributary, most u/s site	1	BA	C	27º25'02" S	120º35'21" E
Northern tributary	2	B*	C	27º25'30" S	120º34'54" E
Northern tributary, near Six-Mile Well	3	B*	C	27º25'52" S	120º34'43" E
Northern tributary	4	BA	I	27º26'58" S	120º34'05" E
Main channel at Old Hwy	5	BA	I	27º27'59" S	120º33'35" E
Main channel at telegraph line crossing, west of Goldfields Hwy	6	BA	I	27º29'39" S	120º30'42" E
Northernmost of SW Claypans (NE of Yakabinda Well)	7	BA	I	27º31'52" S	120º26'57" E
Northernmost of SW Claypans	Depth gauge	BA	I	27º31'53" S	120º26'53" E
Eastern tributary, near confluence with main channel	8	BA	I	27º27'53" S	120º33'48" E

Codes: * Site situated in area of future mine pit and to be replaced by additional upstream control sites as pit expands. B = site surveyed before commencement of mining; A = site to be re-surveyed after commencement of mining; C = control/reference site; I = exposed site (potentially impacted by mining).

2.3 Riparian Condition

Riparian vegetation is defined as vegetation on any land that adjoins or directly influences a water body. Though riparian zones typically only occupy a small proportion of the landscape, they are usually the most productive, with higher species richness and abundance. Increased shade, shelter and lower temperatures provided by taller trees fringing the channel provide habitat for terrestrial fauna such as birds, reptiles and native mammals. Leaf litter and woody debris falling into the creeks offers in-stream habitat and food sources for aquatic invertebrates and (where present) fish. Macrophytes (*e.g.* rushes, sedges) provide habitat for aquatic and semi-aquatic macroinvertebrates. The open waters along creeks and pools frequently provide foraging habitat for bats. Loss of these zones leads to loss of biodiversity and can have widespread ramifications downstream; *e.g.* increased nutrient and sediment loads and increased surface runoff resulting in flooding and channel erosion.

Brief assessments of local riparian vegetation condition were made on the basis of dominant plant species and relative degree of disturbance such as weed invasion, livestock access and fire *etc*. The proportion of exposed soils along banks at each site was also estimated. Assessments were broadly based on the rapid assessment methodologies of Pen and Scott (1995) and WRC (1999) developed for south-west rivers, acknowledging that these approaches are not directly applicable to temporary watercourses in the Goldfields.

Riparian zones are topographically unique and within the study area they included:

- the land immediately alongside the channel,
- land that may be periodically connected to the channel *via* surface runoff (*e.g.* gullies and dips) and
- claypans and playas on the creek floodplain which may periodically interconnect with the creek.

2.4 Physico-chemistry

Measurements of water quality were made in conjunction with the fauna sampling. Parameters recorded are summarized in Table 2. Measurements of temperature, dissolved oxygen, conductivity and pH were made *in situ* between the hours of 0800 and 1800 using portable WTW field meters. A Secchi disk was used for field determination of water clarity. Undisturbed water samples were taken for laboratory analyses of turbidity, colour, ionic composition, metals and nutrients. Samples collected for nutrients and metals were filtered through 0.45 μm Millipore nitrocellulose filters and acidified in the field. Nitrates and soluble reactive phosphorus were recorded as an indication of the bio-available component of phosphorus in the water column.

Table 2. Physico-chemical parameters measured at each site.

Parameter	Code	Units
Water depth	Depth	cm
Wetted width	Width	m
Dissolved oxygen	DO %	% saturation
Dissolved oxygen	DO mg/L	mg/L
Electrical Conductivity	ECond	μS/cm
pH	pH	pH units
Water Temperature	Temp	°C
Colour	Colour	TCU (True Colour Units)
Turbidity	Turb	NTU (Nepholemetric Turbidity Units)
Water clarity	Secchi	cm
Sodium	Na	mg/L
Calcium	Ca	mg/L
Potassium	K	mg/L
Magnesium	Mg	mg/L
Chloride	Cl	mg/L
Carbonate	HCO_3	mg/L
Sulphate	SO_4	mg/L
Arsenic	As	mg/L
Cadmium	Cd	mg/L
Chromium (unfiltered – as tota)	Cr	mg/L
Copper	Cu	mg/L

Parameter	Code	Units
Lead	Pb	mg/L
Nickel	Ni	mg/L
Mercury	Hg	mg/L
Selenium	Se	mg/L
Zinc	Zn	mg/L
Nitrate	NO_3	mg/L
Total soluble nitrogen	TN	mg/L
Soluble reactive phosphorus	P-SR	mg/L
Total soluble phosphorus	TP	mg/L

Only single samples from the largest of the pools along each 200 m sampling reach were taken. All samples were stored on ice in the field and 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 was assessed against the water quality guidelines for the protection of aquatic ecosystems (ANZECC/ARMCANZ 2000a).

Water depth was measured using a graduated pole. Dominant habitat substrates were visually appraised (by surface area) for mineral or other (*e.g.* vegetation, organic detritus) material. Extent of bank erosion and channel down-cutting was qualitatively assessed. Descriptions of overall stream condition were based on categories outlined in WRC (1999).

2.5 Aquatic Invertebrates

At each site, two invertebrate sub-samples were collected; 'benthic' samples targeting macroinvertebrates and 'planktonic' samples targeting micro-invertebrates. Areas sampled for macroinvertebrates were selected to represent a range of habitat types (*e.g.* open water, rushes, fringing/draped vegetation, root mats, woody debris, gravel/sand substrate *etc*) and maximise the diversity of species collected at each location.

2.5.1 Macroinvertebrate Fauna – Benthos

At each site, sampling was conducted with a 250 µm mesh pond-net to selectively collect the macroinvertebrate fauna over a total 50 m of pool habitat within each 200 m reach of stream bed. This sampling protocol was based upon a standardised sweep-net approach adopted for previous surveys of this type (see Storey *et al.* 1993, Edward *et al.* 1994, Halse & Storey 1996). Streamtec (1992) noted that macroinvertebrate diversity in Jones Creek appeared dependent on pool size. In order to ensure valid between-site and between-year comparisons, current sampling was therefore conducted over adjacent river pools to give the total 50 m sweep. Thus sites, *per se* were not fixed but rather representative of a reach, given that channel pools were likely to 'move' during high flows, due to the highly mobile sand bed. Subsequent sampling will be of representative pools in the approximate vicinity of current site locations.

All identified habitats ≤ 1 m deep were sampled including water column, submerged vegetation, bottom sediment, along submerged logs and around tree trunks. Bed substrates were vigorously disturbed with repeated sweeping. Contents of the pond-net were emptied into a bucket several times during sampling to reduce resistance of the net in the water. The nets were thoroughly rinsed between sites and air-dried to eliminate the risk of transfer of material between sites and to allow for any specimens accidentally transferred into subsequent samples to be identified (desiccated) and excluded from analysis.

Bulk samples were preserved in 70% alcohol and returned to the laboratory where each was separated into three size fractions using 2 mm, 500 μm and 250 μm sieves. Representatives of each species (or morphospecies) were picked out using a dissecting microscope with 10 – 50x magnification. All taxa were identified to the lowest possible level (species, where possible) and enumerated to $\log_{10}$ scale abundance classes (*i.e.* 1 = 1 to 10 individuals, 2 = 11 to 100 individuals, 3 = 101-1000 individuals, 4 = 1001^{+}). Given the nature of this survey, it was considered that $\log_{10}$ scale abundance classes were adequate for detecting major changes in the fauna (*e.g.* change in relative abundance) and are a good cost-compromise over counting all animals, particularly as a quasi-quantitative sampling protocol was used.

2.5.2 Micro-invertebrate Fauna - Zooplankton

For the purposes of this study, the micro-invertebrate fauna was defined as all micro-crustacea (Copepoda, Ostracoda, Cladocera), rotifers and Protozoa. Previous studies have shown that for assessing the conservation value of aquatic systems in semi-arid and arid Australia the rarer components of the aquatic invertebrate fauna tend to be in the micro-invertebrate fauna (Storey *et al.* 1993, Halse & Storey 1996, Halse, Storey & Shiel unpub. dat.). However, this may partly reflect sampling effort as the micro-invertebrate fauna historically has been less studied and therefore tends to produce records of new taxa. In this study, micro-invertebrate assemblages were sampled with a 53 μm mesh net over a total 50 m using the same sampling protocol as macroinvertebrates. However, benthic sediments were not disturbed. All samples were preserved in 4% buffered formalin for several days, before being transferred into 70% ethanol. Micro-invertebrate samples were processed by picking the first 200 – 300 individuals from an agitated sample (20 - 30 ml aliquot) decanted into a 125 mm^2 grided plastic tray, with all individuals identified to species, with the tray then scanned for additional missed taxa also taken to species. Each individual plankter/microfauna encountered was scored on a multi-channel Micro-Professor tally counter, identified *in situ*, or removed for erosion (rotifers) or disarticulation (micro-crustacea) on a Zeiss SV-11 dark-field dissecting microscope. This method produces a qualitative estimate of abundance based on species recorded in the 20 – 30 ml sub-sample. Owing to the extremely high numbers of individuals per sample, it is not economic to process entire samples.

2.6 Fish Fauna

Though fish were not expected to occur in the episodic creek and claypans, surveys were conducted to confirm their absence. Baited mesh box traps were set overnight in the deeper channel pools. Traps were cleared the next morning. Tadpoles and Coleoptera caught in the traps were retained for later identification. Along Jones Creek, the lack of macrophytes and small, shallow pools with waters of high visibility also enabled fish surveys to be conducted by direct observation.

2.7 Tadpoles

Read (1999) noted that frogs may be of particular ecological importance in arid ecosystems given their often high abundance and as such may prove useful as bio-indicators of mining impacts. Therefore, qualitative sampling for tadpoles was conducted during the current study. Sampling was by random sweeps of a 1 mm mesh pond-net and baited mesh box traps used to sample fish. Between 10 and 15 individual tadpoles were retained from each site and preserved in 4% buffered formalin for later identification. Any adult frogs accidentally collected in sweeps were photographed, identified and returned live to the creek.

2.8 Data Analyses

All data, including taxonomic lists, abundance data, physico-chemical measurements and GPS locations were entered onto a Microsoft 'Excel' spreadsheet and a copy lodged with the contracting organisation for future reference. The occurrence of each species as collected by all sampling methods was tabulated. Initial analyses documented species distributions, identifying any species listed as threatened or with limited distributions or restricted to specific habitats and any taxa discovered as new to science. The existence of rare, restricted or endemic species was determined based on the authors' own knowledge and by cross-referencing taxa lists with the Department of Conservation and Land Management (CALM) specially protected and priority species lists and the 2004 IUCN Red List of Threatened Species (IUCN 2004).

All taxa were then coded as either 'Permanent' or 'Temporary' residents depending upon whether they possessed drought-resistant stages (*i.e.* spores, eggs, means to dehydrate or protect themselves from dehydration) and so are always present in the system, or lack such abilities and need to reinvade each time the system flows. Generally, this separated the microfauna, larger crustaceans and some macroinvertebrates (*i.e.* larva of the chironomid midge *Paraborniella tonnoiri*) from all other macroinvertebrates without drought-resistant life stages. Analyses were then performed individually on 'Permanent', 'Temporary' and total fauna (permanent and temporary residents combined).

For interpretation of analyses, fauna were considered as 'rare', 'common' or 'abundant', as log10 abundance categories used for macroinvertebrate are not directly comparable to less qualitative estimates of micro-invertebrate abundances (refer Section 2.5.2 above).

2.8.1 Univariate Analysis

Using sites as replicates, one-way analysis of variance (one-way ANOVA) was used to test for significant differences in community parameters (*i.e.* species richness) and physico-chemical parameters between control and exposed sites.

2.8.2 Multivariate Analysis

Multivariate pattern analysis was performed using ordination techniques to group sites according Permanent, Temporary and total faunal assemblages on both presence/absence and abundance data sets. This approach indicates groups of similar/dissimilar sites based on fauna. Data were analyzed using multivariate procedures from the PRIMER (v5) software package (Clarke & Gorley 2001). Up to four levels of analysis were applied to the data:

1. Describing pattern amongst the fauna assemblage data using cluster and ordination techniques based on Bray-Curtis similarity matrices. The clustering technique uses a hierarchical agglomerative method where samples of similar assemblages are grouped and the groups themselves form clusters at lower levels of similarity. A group average linkage was used to derive the resultant dendrogram. The ordination method used was Multi-Dimensionsal Scaling (MDS) (Clarke & 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.* exposed versus control), 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 (*e.g.* control *vs* exposed) 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:
   - For visualization, the numeric value of key environmental data were superimposed onto MDS ordinations, as circles of differing sizes – so-called 'bubble plots'.
   - The BIOENV routine was used to calculate the smallest subset of environmental variables explaining the greatest percentage of variation in the ordination patterns based on fauna.

Environmental data were analysed using Principal Components Analysis (PCA) ordination to discern patterns, gradients and similarities in water quality amongst the sampling sites. PCA transforms a number of (possibly) correlated variables into a (smaller) number of uncorrelated variables called *principal components*. The first principal component accounts for as much of the variability in the data as possible and each succeeding component accounts for as much of the remaining variability as possible. The starting point for a PCA is a correlation matrix based upon Euclidean distance.

PCA ordinations were performed using untransformed data. For metal concentrations, below the detection limit data were assigned values of half the detection limit for all analyses.

For visualization, the numeric value of key environmental variables was superimposed onto the PCA ordinations, as 'bubble plots'.

2.8.3 Power Analysis and Sampling Design

Statistical power is a means to calculate the sample size (*viz.* number of replicates) needed to detect a given change in a parameter and it will indicate the probability that the sampling regime can actually detect a difference (*viz.* an effect) if one exists (Sheppard 1999). In its simplest form, where there are two groups of samples (*i.e.* upstream control versus downstream exposed, or downstream exposed sampled before and after an event), power analysis shows whether the data are capable of detecting an ecologically important difference in the mean values of the two samples. If the test (*i.e.* t-test or ANOVA) shows a significant difference between the two samples (*i.e.* $p < 0.05$), then by inference, the test is rigorous and has adequate statistical power to detect the existing differences. But, if the test shows no significant difference, this could mean that there actually was no underlying difference between the samples (and this is what is commonly concluded). However, it could also mean that i) there were too few replicate samples to detect the difference, ii) the difference between the means of the two samples (the effect size) might have been real but small, or iii) that the variance in each sample was too large. Power analysis shows the probability that the test could have shown a difference were there to have actually been one.

When designing a monitoring programme, the programme must be able to reliably detect environmental change (*i.e.* significant changes above or below accepted criteria) but also it must be affordable. This is one of the more complex aspects of designing an environmental monitoring programme and is referred to as statistical decision theory (Williams & Choo 1994). The process considers issues such as power analysis (*i.e.* the ability to detect a significant difference, should one exist) and Type I (*i.e.* the mistake of concluding an impact has occurred when it has not) and Type

II errors (the probability of concluding that no impact has occurred even though one has) (see Table 1, after Fairweather 1991). The ultimate aim of an adequate design is to maximise statistical power and therefore minimise the probability of making a Type II error.

Table 3. Statistical outcomes in relation to detecting environmental impacts through a hypothesis-testing approach (after Fairweather 1991)

	Prediction or conclusion of study	
Real state of nature	IMPACT	NONE
IMPACT	Correct	Type II error
NONE	Type I error	Correct

When we use an alpha (α) value of 0.05 (*i.e.* a test must have a p-value <0.05 before we accept statistical significance) and perform one hundred similar experiments using pairs of samples and where we somehow know that there really is no difference between them, then the p-value from the statistical test will fall below this α value about 5 times out of the one hundred. Therefore, we make an error in these five cases – this is a 'Type I error' and is an unfortunate attribute of statistics. We could reduce the chance of making such an error if a more strict α value was used. For example, if we use an α value of 0.01, then we would require $p < 0.01$ for significance. Then, only one in one hundred tests would be significant by chance. But, the difference between the pair of samples would have to be greater before we could achieve a p-value less than 0.01.

There is, however, a complementary problem. If the calculated p-value is greater than the α value (*i.e.* $p > 0.05$), then we accept that there is no significant difference between the two samples. But, if in reality there was a difference and the design was inadequate to detect the difference, then we have made the opposite kind of error, a 'Type II error'. The probability of making this type of error is called Beta (β). Statistical power is referenced as $1 - \beta$, or the probability of being correct when stating that there is no difference between the two samples.

The problem with statistical power and sampling programmes is that if the sampling regime had too few replicates, the test may not be adequate to detect a difference that in reality is there. The smaller the sample, or the smaller the true difference if it exists, the greater is the probability of making a Type II error and this probability increases with increasing variance in the samples. This has great relevance to biological monitoring programmes. For example, if there is a small decline in biodiversity at an important site, or if there is a rise in the contaminant level at a site, then a test with low statistical power may lead us to accept there was no significant difference or no deterioration when in fact there was.

Fairweather (1991) comments that basically, it would be unfortunate if a manager was advised that there has been a significant impact, when in fact there has not (*i.e.* a Type I error was made), but in most cases this type of mistake will be revealed by further investigation and remedial action will either not be initiated or will be suspended. The situation is more serious if a manager is advised that there is no impact, when in fact there is (Type II), and no action is taken! The danger of a poorly designed monitoring programme is that there will be insufficient replication (that is to say, statistical power) to detect a significant impact, should one occur and that the impact will go unnoticed (see Figure 2). An adequate design needs to have sufficient power to detect the impact.

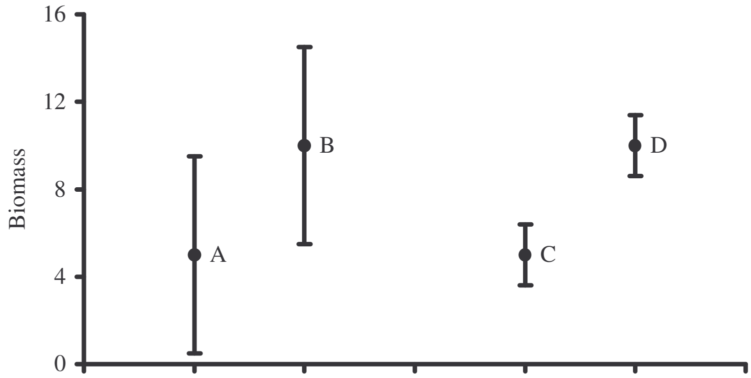


Figure 2. Relationship between sample size, variance and ability to detect a significant difference ($p < 0.05$ & $\beta = 0.95$) for two hypothetical situations, where each data point represents a mean ± 95% confidence intervals. Significant difference is not detectable between populations A and B because of high variance and low replication (n=3). Significant difference is detectable between populations C and D because replication has been increased (n=10).

Fairweather (1991) highlights three applications of power analysis in environmental monitoring: a) for guiding investigators to select the most sensitive statistical tests to apply to the data, b) to indicate sample size (replication) required to detect an impact, based on a pilot study or existing data with known variance and c) to evaluate any non-significant results of completed studies to determine their sensitivity, realised power and detectable effect size. For example, analysis may demonstrate that the design of a programme was inadequate because the programme could only detect a very large change between the two samples (*i.e.* >200% effect size).

Common factors which make a test unable to detect a change include too few samples, too small a difference between the two means, and a large variation in the values making-up the means (Sheppard 1999). Thus, it is when a test does not show a difference that power analysis is especially important. It is needed to show whether or not the test could have shown a difference where difference existed in reality. The ideal is to use power analysis before designing a sampling programme, but it is equally valuable in a post-hoc way, where the data can be analysed, using the means, standard deviations and sample sizes to calculate statistical power.

For designing an on-going monitoring programme for Jones Creek, power analysis was used in both forms:

1. as *post-hoc* tests to calculate the power of tests previously performed on existing data collected in 2005 and
2. as *a priori* tests in calculating sample sizes required when designing the on-going sampling programme.

When undertaking power analysis of the Jones Creek data, several decisions were made in relation to α, $1-\beta$ and effect size. Historically, the statistically accepted α - value of 0.05 has been used and analyses in designing a revised programme adhered to this convention. With respect to the power of a test, the emerging convention is that at a minimum, power should be equal to at least 0.8 (Cohen 1988), or for more rigorous programmes such that $\beta = \alpha$ (*i.e.* if $\alpha = 0.05 = \beta$, then power = 0.95) (Fairweather 1991). The latter value of power of 0.95 is considered stringent and therefore the more conservative target value of power of 0.8 was adopted for these analyses. The final decision to be made related to effect size.

Effect size is a measure of the difference between two samples (*i.e.* a 25% change in a parameter). For the Jones Creek monitoring data the parameter of interest is a specified change in species richness between control and exposed sites at an instance in time, and a change at exposed sites over time. The changes were assessed for total fauna, Permanent fauna and Temporary fauna. The selected effect size basically expresses how big (or small) a change needs to be before it is detected. For a specified effect size, it is then possible to calculate the sample size required to detect the difference, maintaining an $\alpha = 0.05$ and power $(1 - \beta)$ of 0.8. Effect size needs to be small enough to reliably detect environmental change (*i.e.* significant changes above or below accepted criteria) but also it must be affordable; the statistical decision theory of Williams & Choo (1994). For designing the Jones Creek biological monitoring programme, a range of effect sizes were selected to assess the sensitivity of the programme (10% - 50% decline in species richness, tested in 10% increments). Testing changes in species richness between upstream control and downstream exposed sites in any sampling event, and at downstream exposed sites between sampling events.

When calculating the ability to detect an effect size (*viz.* statistical power), the observed data from each set of replicates were statistically manipulated to simulate decreased species richness from 10% - 50% (in 10% increments), and the observed and hypothetical data used to calculate power. Using the observed number of replicate samples taken, data were analysed to determine the number of replicates that would be required to achieve the optimum statistical power of 0.8.

Design of the Jones Creek monitor programme was based on:
$\alpha = 0.05$
power = 0.8
Effect size = 10 - 50% declines in species richness tested in 10% increments
Differences between Control and Exposed sites on any occasion
Differences at downstream Exposed sites between occasions
Effects tested for total fauna, resident fauna and non-resident fauna

3. RESULTS & DISCUSSION

3.1 Catchment & Riparian Condition

Jones Creek catchment was considered extremely degraded due to historic pastoral practices and unrestricted livestock access to the natural waterbodies. Plates 1 - 12 illustrate the conditions at all sites at the time of sampling. Channels at all sites were characterised by extensive erosion, with bank slumping, channel widening and bed down-cutting. Site 2, on the main channel was particularly down-cut with bank height along the outer meander bend reaching 3 m. Banks along the eastern tributary near the confluence with the main channel (Site 8) and in the main channel at Site 6 (west of Goldfields Hwy; Plate 7) were eroded down to the underlying coffee rock ('Murchison cement') with obvious erosion rills on the adjacent flood plain and braiding and terracing of drainage lines. Bed substrates in the creek were dominated by highly mobile, unconsolidated sands (Plates 1, 6 & 11). Wide, deep, lateral sand bars and sand waves were obvious in the channel in the mid and lower reaches, *e.g.* immediately upstream of the Goldfields Highway road bridge (Plate 6) and upstream of Site 8 along the eastern tributary (Plate 11). Channel condition reflected the effects of high and rapid run-off; likely a combination of recent (past ~ 10 years) rainfall events and loss of vegetative cover in the catchment.

The erosive power of a system increases disproportionately with its discharge, thus 1:100 year floods or runoff events are extremely important in forming landscapes. Such floods and events carry the largest quantities of sediment and nutrients. Prior to European settlement, natural vegetation provided a high level of resistance to flows. The clearing of vegetation through rural development and pastoral activities has made river systems sensitive to flooding, to the extent that ARI 1:10 year or similar sized floods may now cause catastrophic erosion (Lovett & Price 1999). Also, down cutting and channelisation often result in increased current velocity and thus also lead to increased bank and bed erosion, increased sedimentation and more severe flooding of downstream reaches (Lovett & Price 1999).

All sites scored an environmental rating of 'fair' to 'poor' indicative of eroding soils and poor vegetation cover (refer Appendix 1). Catchment vegetation was dominated by sparse to open mulga shrubland on hardpan/wash plains, with partly inundated open, Kurara thickets fringing the claypan. Away from the main creekline and claypan, vegetation graded into hummock grass sandplains or sparse, mixed mulga-eremophila shrublands on "stony hardplain" (*sic* Burnside *et al.* 1995) at higher elevation. At all sites, understorey perennials were, at best, sparse with much exposed and apparently 'sealed' soils and only patchy cryptogram (thin layers of algae, lichens, mosses) crusts on top of banks and floodplain.


Plate 1. Site 1, uppermost site on main channel of Jones Creek: (left) mobile sand bed of the channel; (right) bank erosion above channel pools clearly visible in right foreground. May '05.


Plate 2. Site 2, main channel of Jones Creek: (left) visible in mid-ground are the scoured roots of *Eucalyptus camaldulensis* growing within the channel; (right) typical isolated channel pools. May '05.


Plate 3. Site 3, main channel of Jones Creek, near Six Mile Well: (left) view upstream along the channel with fringing eucalypts and mulga; (right) one of the larger, deeper channel pools. May '05.


Plate 4. Site 4, main channel of Jones Creek: (left) broad, open channel with isolated pools; (right) close-up of pool habitat. May '05.


Plate 5. Site 5, main channel of Jones Creek at Old Highway crossing: (left) view upstream along relatively large water body with fringing *Eucalyptus camaldulensis*; (right) view downstream from causeway. May '05.

Plate 6. Immediately upstream of Goldfields Highway road bridge. Highly mobile sand substrates and periodic high energy flows have formed sand waves within the main channel of Jones Creek. May '05.

Plate 7. Site 6, main channel of Jones Creek, west of Goldfields Highway: (left) typical shallow channel pools; (right) eroded and terraced banks with exposed coffee rock. May '05.

Plate 8. Jones Creek near northernmost of the south-west claypans; July '05 (photo K. Shugg).

Plate 9. Northernmost of the south-west claypans; July '05 (photo K. Shugg).

Plate 10. Near depth-marker in northernmost of the south-west claypans; July '05 (photo K. Shugg).

Plate 11. Mobile sand dune/wave within the channel of the eastern tributary, immediately upstream from Site 8. May '05.

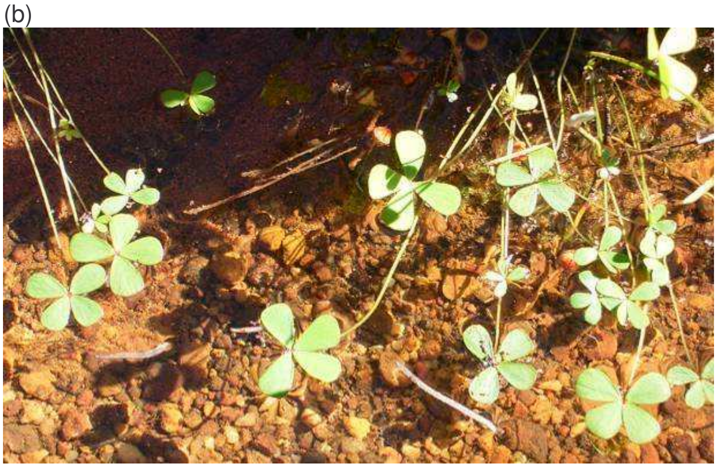
Plate 12. Site 8, eastern tributary of Jones Creek, just upstream from the confluence with the main channel: (left) typical isolated channel pool formed in sandy bed substrates; (right) mulga growing within the channel. May '05.

An extensive, dense, vivid green layer of moss and bryophytes was present mid-way down the banks of the creek (Plate 13a). This layer and the cryptograms on the floodplain likely play an important role in helping to protect fragile soils and absorb nitrogen from the air (Burnside *et al.* 1995) enriching otherwise nutrient-poor sediments. River red gums lined the banks of the main channel, forming a sparse to open woodland providing shade and a limited source of leaf litter and woody debris to the creek. The extent of eucalypt woodland along the eastern tributary was limited to approximately 200 m upstream from the confluence with the main channel; patterns in distribution likely reflecting a combination of the extent of wetted alluvial soils, lateral extent of alluvial soils up the tributary and availability of moisture in dry periods to support the trees. Most tree species observed were mature with little evidence of recruitment along the creek or at the claypan. Heavy scouring of tree roots was prevalent at all creek sites. There was a paucity of in-stream macrophytes, with only isolated clumps of the aquatic fern Nardoo (*Marsilea* sp.; Plate 13b) at the waterline and emergent sedge (*Juncus* sp.; Plate 13c) in pools of Jones Creek. Both are perennial species. Algal cover also appeared low (< 5%). Relatively little is known about macrophytes from intermittent water bodies in arid regions (Porter *et al.* 2001), though Nardoo is known to be common in inland arid areas subject to inundation (Mitchell & Wilcox 1994). The sporocarps (spore capsules) of Nardoo can remain viable for years even under drought conditions.

The local and regional ecological value of the remnant riparian vegetation was considered to be high, in particular the eucalypts, as they represent the only large tree species within the catchment.

(a)

(b)



(c)

Plate 13. In-channel vegetation of Jones Creek: (a) bryophyte and moss layer characteristic of most sites; (b) the aquatic fern Nardoo *Marsilea* sp. at Site 3; (c) emergent macrophyte *Juncus* sp. at Site 3.

3.2 Physico-chemistry

The physico-chemical characteristics of each site are detailed in Appendix 1. Weather conditions over most of the sampling period consisted of moderate cloud cover, occasional light showers during the day, relatively warm air temperatures and light winds. The system was characterised by medium (up to 100 m long x ~10 m wide x ~1.5 m deep at maximum) to small (~ several metres long x wide, with < 0.5 m depth of water) pools in the Jones Creek channel and at the confluence with the southern tributary. The only flow (estimated at around 1 – 2 L/sec) noted was in the main channel, immediately upstream of the confluence with the eastern tributary. This flow was considered the result of recent infiltration of rain on ranges/valley sides and lateral movement through the solid profile downslope into the valley floor, percolating into the creekline. The presence of an extensive moss & bryophyte layer mid-way along the creek banks appeared further evidence of this lateral seepage. All other sites were standing-water pools, however, subsurface flow was likely present through the saturated alluvial deposits.

Maximum water depth in the northernmost claypan, south-west of the Highway could not be determined but likely exceeded 2 m (Doug & Lucy Brownlie, pers com.). Water level at the depth gauge was 1.36 m at 1000 hrs on 19th May 2005, determined by ability to wade into the claypan and securely insert a temporary gauge.

Values for most of the physico-chemical variables measured were within the ranges expected for 'slightly-moderately disturbed' ecosystems of north[7] Australia (ANZECC/ARMCANZ 2000a; refer Appendix 2). Waters in Jones Creek were clear but slightly tannin-stained, while those of the claypan were very turbid (Secchi 6.0 cm; turbid. 320 NTU). Turbid water is typical of claypans throughout Australia and is generally considered a function of water depth rather than stage in the hydrological cycle (Timms 2002). Water temperatures ranged from 18.5 – 22.3 °C, likely reflecting differences in water depth (*i.e.* shallower wetlands have greater daily ranges in temperature), time of day when sampled (wetlands sampled in the morning will likely be cooler than those sampled in the afternoon), and turbidity (water with high percentage of suspended solids tends to absorb more heat than clear water). Dissolved oxygen levels were unexpectedly high at many sites ranging from 74% to 144% (super-saturated). Conductivity ranged from 52.0 - 132.1 µS/cm, reflecting very fresh water. Water pH ranged was 5.1 - 6.8, with lowest pH occurring at Site 5 at the old highway crossing. The low conductivity and circum-neutral pH at most sites, was likely due to the direct influence of rainwater. The reason for the lower pH values recorded at Site 5 could not be determined.

The composition of major ions was noticeably different in the claypan compared with Jones Creek sites; the claypan was dominated by potassium and bicarbonate ($K^+ > Na^+ > Ca^{2+} > Mg^{2+}$: $HCO3^- > Cl^- > SO_4^{2-}$) whereas the creek sites were dominated by sodium/calcium and bicarbonate ($Na^+ <> Ca^{2+} > K^+ \geq Mg^{2+}$: $HCO_3^- > Cl^- > SO_4^{2-}$). The ionic composition of waters within the catchment will be determined by rain-borne salts (*i.e.* wind blown dusts) and geology (*e.g.* weathering of soils) of the catchment. However, the composition, particularly in shallow waters (*e.g.* claypans) will be altered by evapo-concentration and precipitation of less soluble salts, such as calcium carbonate and magnesium sulphate (Hart & McKelvie 1986). The higher contribution of potassium in claypan waters may reflect weathering of potassium-bearing feldspars. Given the relatively high bicarbonate content (60 - 70% of the total major anion concentration) in creek waters, it might be expected that waters would be alkaline however, this was not the case. Elevated bicarbonate is a characteristic of many fresh inland waters (Hart & McKelvie 1986); dominance or sub-dominance of Ca^{2+} and HCO_3^- reflecting the importance of weathering. The ionic composition of inland waters in Australia is known to vary widely.

Total nitrogen concentrations ranged from 0.46 - 2.4 mg TN/L, nitrates ranged from 0.06 – 1.9 mg NO_3/L and total phosphorus 0.02 to 0.06 mg TP/L. Soluble reactive phosphorus (SRP) was at or below the detection level of 0.01 mg/L for filterable reactive phosphorus (FRP). TN, NO_3 and TP concentrations exceeded ANZECC/ARMCANZ (2000a) guideline values at all sites. Nitrogen levels were particularly high at Site 6, west of Goldfields Highway. Maximum TP concentrations were recorded from the uppermost control site (Site 1), along the northern tributary. The elevated nutrient concentrations recorded were not unexpected, given the past land management practices and unrestricted access of livestock to the natural water bodies.

Classifying pools to a trophic status is a useful means to summarise nutrient data in that it provides an indication of current status, allows comparisons amongst sites and may be used to assess change in a site over time. There are a number of classification schemes whereby waters may be classified to a trophic status based on nutrient concentrations (*e.g.* Wetzel 1983, OECD 1982, Salas & Martino 1991). The classification by Wetzel (1983) is from the classic text on

[7] Data were compared against ANZECC/ARMCANZ (2000a) water quality guidelines for the protection of ecosystems, using trigger values specific to slightly disturbed wetlands of north-west and tropical Australia (refer Appendix 2), as these were considered most applicable, given the streamflow and rainfall patterns of the study area.

limnology based on temperate northern hemisphere lakes, and uses ranges in concentrations of TP, TN and inorganic N (Table 4). The OECD classification is based on a survey of a large number of lakes, mainly from northern temperate areas, particularly glacial lakes from high latitudes. The system was produced by qualitatively classifying each lake to a trophic state, and then using measurements of nutrient concentrations and loads, chlorophyll-a and Secchi disc transparency to set boundary values and the mean and variance of each parameter for each trophic category (Davis *et al*. 1993).

A similar approach was used by the Pan American Centre for Sanitary Engineering and Environmental Sciences (CEPIS, Salas & Martino 1991) to develop a classification system for warm-water tropical lakes (Table 5). Forty lakes were individually assigned to a trophic state based on qualitative criteria, and site-measurements of mean annual total phosphorus were then used to determine standards (means ± standard deviations) for each category. For the CEPIS study, warm-water lakes were defined as lakes with a minimum temperature of greater than 10 ºC, with a minimum annual average of 15 ºC. All pools sampled in the current study were considered to be warm-water sites and therefore, the classification by CEPIS is most appropriate.

There are currently no WA or Australian-specific guidelines for determining the trophic status of freshwater bodies. OECD and CEPIS do not classify lake trophic status according to TN. However, according to Wetzel (1983), the majority of sites in this study classified as eutrophic on TN concentrations. Site 6 had TN above the upper limit for hyper-eutrophic. Spot measurements of TP indicated most sites were within the mesotrophic range according to the CEPIS classification (Table 5).

Table 4. Classification of lake trophic status based on ranges in nutrient concentration (Wetzel 1983).

Category	Total P & Ortho P (mg/L)	Total N (mg/L)	Inorganic N (mg/L)
Ultra-oligotrophic	0 – 0.005	0 – 0.25	0 – 0.20
Oligo-mesotrophic	0.005 – 0.01	0.25 – 0.60	0.20 – 0.40
Meso-eutrophic	0.01 – 0.03	0.30 – 1.10	0.30 – 0.65
Eutrophic	0.03 – 0.1	0.50 – 1.50	0.50 – 1.50
Hyper-eutrophic	> 0.1	> 1.50	> 1.50

Table 5. Mean annual total phosphorus levels (mg/L) for each trophic category under the OECD (1982) and CEPIS (Salas & Martino, 1991) classifications (means and range over 2 standard deviations).

	Category		
	Oligotrophic	Mesotrophic	Eutrophic
OECD	0.008	0.0267	0.084
(± 2 SD)	(0.003 – 0.022)	(0.008 – 0.091)	(0.017 – 0.424)
CEPIS	0.021	0.0396	0.1187
(± 2 SD)	(0.010 – 0.045)	(0.021 – 0.074)	(0.028 – 0.508)

Most metal concentrations in the water column were below ANZECC/ARMCANZ (2000a) guidelines with the exception of chromium, copper, nickel and zinc. Total chromium in the waters of the claypan (0.017 mg/L) was much greater than the guideline level (0.001 mg/L) for Cr (VI). Tests (*e.g.* DGTs) should be performed to determine if high total Cr levels are indicative of high levels of the more toxic Cr (VI) and the bioavailability of these metals. Creek sites 2 and 6 had total Cr levels of 0.002 – 0.003 mg/L with all other sites <0.002 mg/L, however it should be noted that detection limits were only 0.002 mg/L. Copper concentrations ranged from 0.003 – 0.007 mg/L and were well above the guideline level of 0.0014 mg/L. Maximum levels were recorded at the old Highway crossing on Jones Creek (Site 5) and in the claypan (Site 7). Nickel was also slightly elevated in claypan waters, with a concentration equal to the guideline level of

0.011 mg/L. In Jones Creek, highest Ni concentrations were also recorded at Site 5 (0.008 mg/L). Elevated zinc concentrations of 0.018 – 0.12 mg/L (*cf* the guideline level of 0.008 mg/L) were recorded at all sites, except creek Sites 3 and 4. Like copper, maximum zinc levels were recorded at Site 5. Though below guideline limits (0.0034 mg/L), lead concentrations were also highest at Site 5 (0.002 mg/L) and the claypan (0.001 mg/L).

SKM (2005) found concentrations of Cr & Ni were also elevated in sediments of the creek and in particular the south-west claypans. While these metal levels should be treated as background levels or 'reference' levels against which to compare any future mine impacts, the extent to which elevated metals reflect past anthropogenic disturbance is not known. The relatively high levels of Cr, Cu and Ni in claypan waters and/or sediments may indicate a tendency for these metals to accumulate here, while the more acidic waters (pH 5.1) at Site 5 may indicate a greater potential for mobilization of some metals from bed sediments. On-going monitoring will indicate if low pH is typical for this site.

3.2.1 Patterns in Physico-chemical Data

Though physico-chemistry varied between individual sites, there were no significant differences (ANOVA; df = 1, F = 0.00393, p = 0.950) between control (Sites 1, 2 & 3) and exposed sites (Sites 4, 5, 6 & 8) for any of the measured environmental variables. Therefore sites selected as controls should serve as good pre-mine reference sites against which to compare any future changes in water quality of the receiving environment. Sites 2 and 3 however, will not be used as long-term controls as they are within the footprint of the orebody (refer Section 2.2.1).

PCA ordinations of physico-chemical parameters showed a clear separation of the claypan from the Jones Creek sites (Figure 3) on PCA component 1 (PC1). PC1 and PC2 accounted for 70% of the total sample variance (47% and 23% respectively). Parameters contributing most to the variance on PC1 were turbidity, Cr, Ni and those associated with composition of major ions (*i.e.* conductivity, Ca^{2+}, Mg^{2+} & HCO_3^-). Major contributing variables on PC2 were pH, Na^+, Cu, Pb and Zn. Slightly higher SRP also contributed to the separation of the claypan from all other sites. Even when the outlier claypan site was omitted from the analysis, there was no clear separation of control from exposed sites (Figure 3). The PCA plot however did show a separation of Site 5 from all other creek sites (Figure 3).

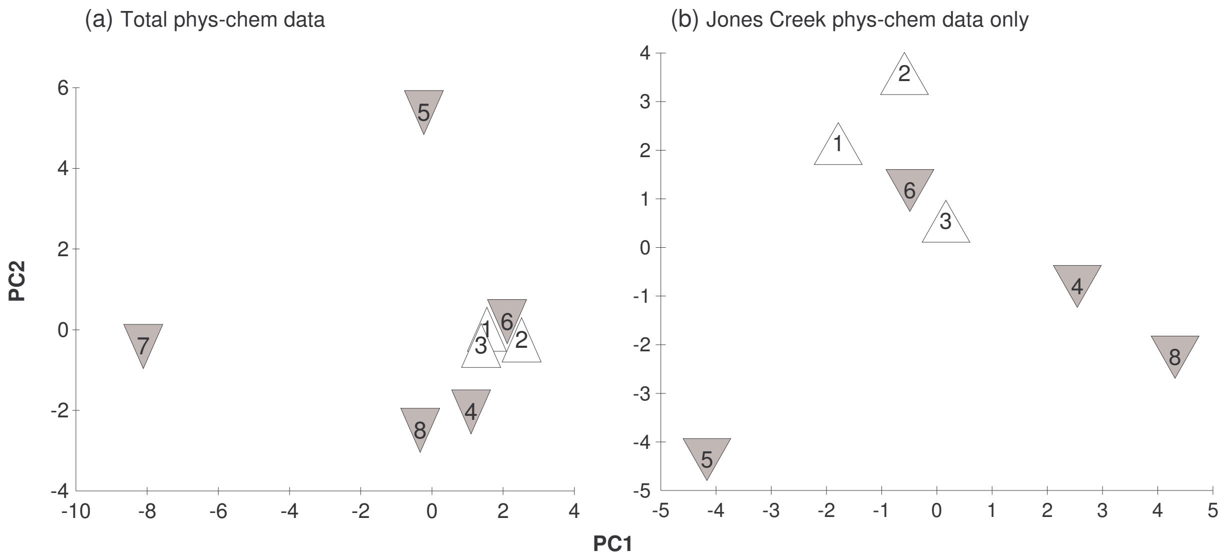


Figure 3. Two dimensional PCA ordination plots of physico-chemical variables for (a) total samples and for (b) creek samples only. Codes: ▼ exposed sites; △ control sites.

PC1 and PC2 explained 62.5% of the creek sample variance (32.1% and 30.4% respectively). Parameters contributing most to the variance were those associated with composition of major ions (*i.e.* conductivity, Ca^{2+}, Na^{+}, Mg^{2+}, HCO_3^- & SO_4^{2-}), dissolved oxygen concentration, pH and metals Zn, Ni, Cu and Pb. As noted above, pH was lower and Cu, Pb, Ni and Zn higher at Site 5 than all other creek sites which likely explains the separation of this site from others in the ordination.

3.3 Aquatic Invertebrate Fauna

3.3.1 Taxonomy

In total, 124 taxa were recorded from the eight sites sampled in May 2005, with fauna dominated by insects (37%), rotifers (33%) and crustacea (17%). Chironomidae (non-biting midges) constituted 28% of all insect taxa. A list of all taxa collected is presented in Appendix 3, indicating which taxa are likely to be 'permanent' residents and which are likely to be 'temporary', rapid colonisers following rain. The taxonomic listing also includes records of larval and pupal stages for groups such as Diptera (two-winged flies) and Coleoptera (aquatic beetles). Current taxonomy in Australia is not sufficiently well developed to allow identification 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. Table 6 provides a summary of the major types of invertebrate fauna collected. No introduced macroinvertebrate taxa were recorded.

Table 6. Summary of invertebrate fauna recorded from Jones Creek and the claypan during May 2005.

Micro-invertebrates	No. of 'species'	Macroinvertebrates	No. of 'species'
Protista (protozoans)	12	Turbellaria (flat worms)	1
Rotifera	41	Oligochaeta (aquatic worms)	1+
Copepoda	8	Mollusca (Gastropoda, ?*Isidorella* sp.)	1
Cladocera (water fleas)	4	Macro-crustacea (clam shrimps, shield shrimps, fairy shrimps)	5
Ostracoda (seed shrimps)	5	Diptera (two-winged flies)	27+
		Lepidoptera (moths)	1
		Anisoptera (dragonflies)	1
		Ephemeroptera (mayfly, *Cloeon* sp.)	1
		Hemiptera (true bugs)	3
		Coleoptera (aquatic beetles)	13
Total number of 'species'	**70**	**Total number of 'species'**	**54**

Of species recorded, 63% were considered to be 'permanent' residents with desiccation-resistant life stages that would allow them to remain within the creek or claypan once surface waters had evaporated. The majority of these species were the micro-crustacea (protists, rotifers, copepods & ostracods) and branchiopods (notostracans, anostracans, conchostracans & cladocerans). Many of which are known to emerge within hours of flooding and develop quickly over a period of about two weeks, before the more predatory colonisers appear. All of the early, 'temporary' invaders were comprised of Insecta.

Most taxa recorded are considered tolerant of a wide range of environmental conditions and are common, ubiquitous and frequently encountered in ephemeral river systems and wetlands within Western Australia. Of note were range extensions for five species of micro-invertebrate which had not previously been recorded within Western Australia. One of these, the notommatid rotifer *Cephalodella* sp. nov. (from Site 3), is a new and as yet undescribed species, previously only known from Lake Eyre catchment in South Australia. Range extensions were also recorded for the lecanid rotifer *Lecane* cf. *eylesi* (Site 1) and notommatid rotifers *Cephalodella euderbyi* (Site 4), *Cephalodella* cf. *pachdactyla* (Site 5) and *Cephalodella ventripes* (all sites except the claypan).

Widely disjunct distributions are not uncommon for rotifers, an example being the discovery by Russ Shiel of a rotifer in a high-altitude pond on Mt Beauty, Victoria, which was previously known only from a similar pond on Mt Desert Island, Maine, USA. These types of translocations are likely attributable to the resting eggs (ephippia) of rotifers, copepods and cladocerans being caught in feathers of migratory birds, thus making long distance dispersal possible. Branchiopod fauna of ephemeral waters also appear to have similarly wide distributions, though Williams (2002) believes that many cladocerans are likely to be geographically restricted.

As per Storey *et al.* (2004), the taxonomy of the non-biting midges, Chironomidae, remains incomplete and various species and complexes (*e.g. Tanytarsus* spp.) are open to conjecture. The *Parachironomus* keyed closest to 'K2 Cranston', which has previously been recorded from the Northern Territory (D.H Edward, UWA, pers. comm.). Similarly *Polypedilum* sp. keyed closest to 'M1 Cranston' known only from Qld, NSW and Victoria. Hence specimens recorded from Jones Creek may represent a range extension or even a new species. *Chironomus tonnori* is known to be an early coloniser of temporary waters, while in contrast, the larvae *Paraborniella tonnoiri* are believed able to survive in a desiccated state in bed sediments, rapidly re-hydrating body tissues once water returns (D.H. Edward, UWA, pers. comm.).

Other macroinvertebrates of interest included the shield shrimp *Triops australiensis australiensis* and the fairy shrimp *Branchinella occidentalis* (Plate 14) which were recorded in high numbers from the claypan but were absent from Jones Creek.

(a)

(b)

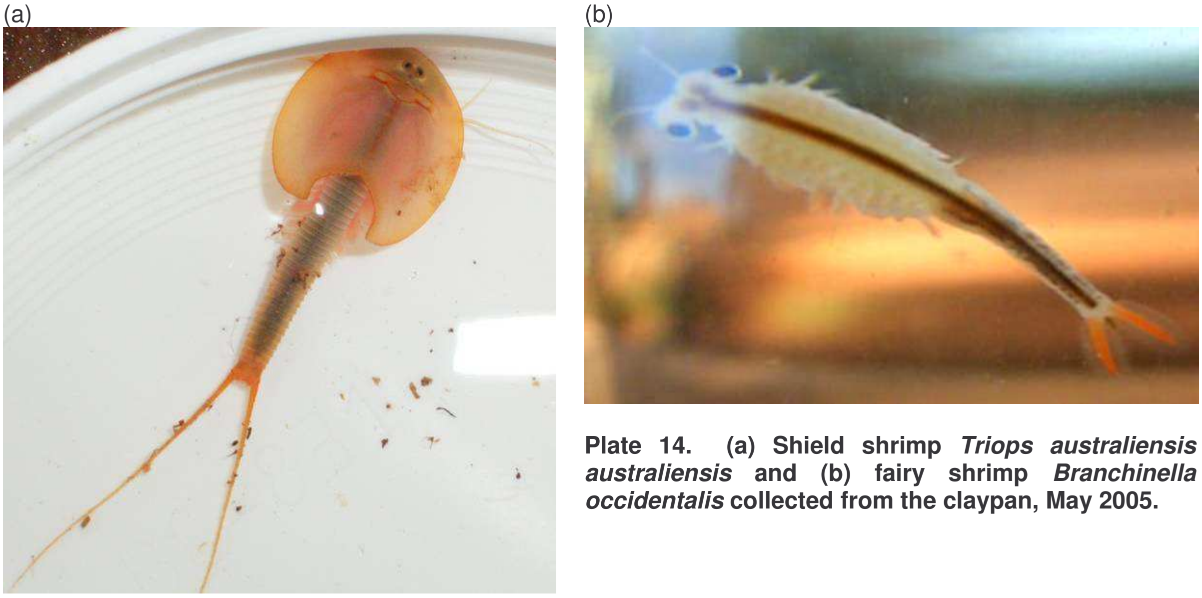


Plate 14. (a) Shield shrimp *Triops australiensis australiensis* and (b) fairy shrimp *Branchinella occidentalis* collected from the claypan, May 2005.

Both species are commonly recorded from claypans across Australia and *T. (a) australiensis* is frequently found in station dams. Both appear largely restricted to lentic waters. The eggs of *T. (a) australiensis* are extremely resistant to drought and may be translocated long distances by the wind (Williams 1980). The external morphology of *Triops* species has remained unchanged since the Triassic period, approximately 200 million years ago and the genus is therefore believed to contain the oldest living animal species on earth.

The only mayfly species recorded, *Cloeon* sp. was here considered a temporary resident of the creek, however, Paltridge *et al.* (1997) speculated that at least some species of *Cloeon* may possess desiccation-resistant eggs.

3.3.2 Taxa Richness

Species diversity ranged from 33 in the claypan to 62 at Site 6 on Jones Creek (Figure 4). One-way ANOVA detected significantly greater species richness in the temporary fauna of exposed sites compared to control sites (Table 7). However, there were no significant differences for permanent resident fauna or total fauna.

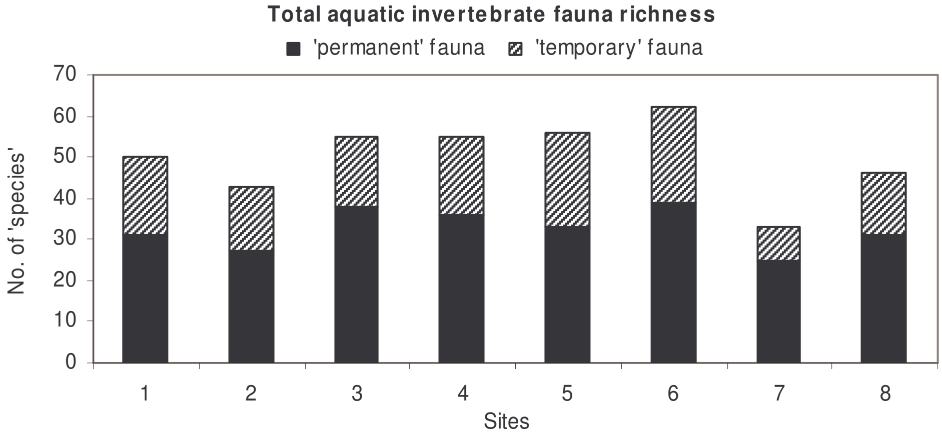


Figure 4. Richness of aquatic invertebrate fauna in Jones Creek and the claypan (Site 7).

Table 7. Summary of one-way ANOVAs on between-control/exposed sites in invertebrate species richness for data collected from Jones Creek in May 2005. Tukey's HSD multiple comparison test was used to locate between-level differences for significant main effects (ns = not significantly different at p < 0.05 and effects are in descending order, with mean values in parentheses.

Effect	df	F-value	p-value	Tukeys HSD Range test		
Total Fauna	1,4	2.954	ns	Exposed (59)	=	Control (51)
Permanent Resident Fauna	1,4	0.852	ns	Exposed (36)	=	Control (32.3)
Temporary Resident Fauna	1,4	10.56	0.0314	Exposed (23)	>	Control (18.7)

Only 4% of taxa were common to all seven sites (refer Appendix 3). These were bdelloid rotifers, the notommatid rotifer *Cephalodella gibba,* cyclopoid copepods and culicids (mosquito larvae). Only 15 % of taxa occurred at more than 80% of sites. Forty-eight taxa (38%) were singletons, occurring at only one site and many of these were species of Protista and Rotifera. The relatively low numbers recorded from the claypan were considered most likely due to the highly turbid waters. Turbidity is known to have considerable effect on community composition (Williams 1998), though Halse *et al.* (1998) suggested ionic composition may also play a role. Higher water velocities and mobile bed substrates will also undoubtedly influence faunal community structure of the creek. As noted above, there were species that appeared restricted to the claypan including the planorbid gastropod ?*Isidorella* sp., the conchostracan *Caenestheria* sp., a number of rotifers (*Lecane* sp. A, *Lophocharis salpina*, *Cephalodella* sp. A & *Notommata* sp. A), *Triops* (a) *australiensis* and *Branchinella occidentalis.* Of interest, the only other anostracan collected *Branchinella frondosa*, was only recorded from Jones Creek. Timms *et al.* (in prep.), similarly reported that of the 19 species of anostracan recorded from the Carey wetlands, most generally live in very turbid waters, with one exception being *B. frondosa*. In the Carey wetlands, *B. frondosa* appeared to favour the more vegetated sites and fresh, clear creek pools. Such specific requirements of fauna in terms of turbidity, water velocity, ionic composition and stable bed substrates have ramifications for the proposed creek diversion which has the potential to affect all these environmental parameters.

There were a number of species which were recorded only from Site 3, located within the footprint of the proposed main pit. Species included four rotifers, one of which was the species new to science (*Brachionus* cf. *lyratus*, *Encentrum* cf. *uncinatum*, *Cephalodella megalocephala* & *Cephalodella* sp. nov.), one cladoceran (*Ceriodaphnia* sp.) and two beetles (*Allodessus bistrigatus* & the semi-aquatic Staphylinidae spp.). It is highly unlikely that any of these species are unique to this one creek site. The beetles in particular are highly mobile 'vagrants'. The apparent restriction of the rotifers and cladoceran was considered an artefact of the sampling and they are likely to be present at all creek sites, but by chance were only recorded in the sub-samples from Site 3.

Halse *et al.* (1998) in similar studies of Lake Gregory - a large (380 km^2), mostly fresh, semi-permanent lake on the edge of the Great Sandy Desert - considered the 174 taxa recorded over three sampling occasions to represent a highly diverse fauna for inland waters. The 124 taxa collected from the Jones Creek catchment in one-off sampling in May 2005, would thus also appear to represent a highly diverse aquatic fauna. Timms *et al.* (in prep.) reported at least 107 taxa of invertebrates recorded over four sampling occasions from saline and fresh wetlands (750+ km^2), in the Lake Carey catchment to the south-west of Yakabindie. However this study did not target rotifers or micro-crustaceans. Amongst other substantially larger, more extensively surveyed arid systems, riverine waterholes of the Lake Eyre basin reportedly support 136 macroinvertebrate species and more than 400 micro-invertebrates (refer Timms *et al.,* in prep.), while the Paroo River supports at least 200 invertebrate taxa (refer Timms & Boulton 2001, Timms 2002).

By contrast, Streamtec (1992) recorded only 68 taxa from Jones Creek in January 1992. The lower taxonomic resolution used (many taxa, particularly micro-invertebrates, were not identified to species-level) likely accounts for the much lower diversity reported in this earlier study. Stage in hydroperiod and associated successional stage in colonising fauna may also account for variations in presence/absence of temporary species such as coleopterans. Streamtec (1992) reportedly identified a number of adult dragonflies, however, larvae of only two odonate species, (*Austrolestes annulosus* & *Orthetrum caledonicum*) were actually collected from waters of the creek. The current study recorded juvenile dragonflies at sites 5 and 6, but individuals were too immature to positively identify.

3.3.4 Multivariate Analysis Fauna

Classification of both permanent and temporary resident fauna samples on presence/absence data clearly showed the separation of the claypan from the creek sites. The classification dendogram (Appendix 4) indicated less than 30% similarity between claypan and creek fauna, therefore claypan samples were effectively outliers and omitted from subsequent between-site comparisons. Analyses of abundance data (not shown) gave a similar result.

MDS ordinations of the species data showed some separation of creek sites, with Site 8 (eastern tributary) tending to group separately on abundance data (*e.g.* Figure 5). However, ANOSIM detected no significant separation of *a priori* groups, *i.e.* control (Sites 1, 2 & 3) from exposed sites (4, 5, 6 & 8) in ordination space. Results were similar for all fauna, *i.e.* permanent, temporary and total fauna.

The relationships between creek fauna data and environmental variables were investigated using BIOENV procedures (Clarke & Warwick 2001). Physico-chemical variables most likely to be influencing community structure within Jones Creek were Cr, Zn, Ni, Na^{+}, nutrients (TN, NO_3 & TP) and turbidity. Examples of three of these are shown as 'bubble' plots in Figure 4.

Pairwise comparisons of invertebrate fauna showed a high percentage similarity between individual creek sites, in particular for presence/absence data for permanent residents (mean 72%; Table 8). The variation that did exist between sites was considered largely due to singletons (species recorded from only one site). Between-site similarity for the control group (Sites 1, 2 & 3) was 71% and equal to that within the exposed group (Sites 4, 5, 6, & 8) (Table 9). Comparisons indicated little difference between Site 8, the sole site on the eastern tributary, and other exposed sites (Table 9), with no significance difference (one-tail t-test, df = 2, t value = 0.45, p = 0.35) in diversity of permanent fauna between Sites 4, 5 and 6 versus Sites 5, 6 and 8.

Table 8. Mean percentage similarity of invertebrate fauna between creek sites

	Presence/absence data	Abundance data
Temporary resident fauna	58	56
Permanent resident fauna	72	70
Total fauna	67	64

Table 9. Mean percentage similarity of permanent fauna within control and exposed groups.

	Control Sites **1, 2 & 3**	Exposed Sites **4, 5, 6 & 8**	Exposed Sites **4, 5 & 6**	Exposed Sites **5, 6 & 8**
Permanent resident fauna	71	71	72	69

Pairwise comparisons of invertebrate fauna showed a high percentage similarity between individual creek sites, in particular for presence/absence data for permanent residents (mean 72%; Table 8). The variation that did exist between sites was considered largely due to singletons (species recorded from only one site). Between-site similarity for the control group (Sites 1, 2 & 3) was 71% and equal to that within the exposed group (Sites 4, 5, 6, & 8) (Table 9). Comparisons indicated little difference between Site 8, the sole site on the eastern tributary, and other exposed sites (Table 9), with no significance difference (one-tail t-test, df = 2, t value = 0.45, p = 0.35) in diversity of permanent fauna between Sites 4, 5 and 6 versus Sites 5, 6 and 8.

Table 10. Mean percentage similarity of invertebrate fauna between creek sites

	Presence/absence data	Abundance data
Temporary resident fauna	58	56
Permanent resident fauna	72	70
Total fauna	67	64

Table 11. Mean percentage similarity of permanent fauna within control and exposed groups.

	Control Sites **1, 2 & 3**	Exposed Sites **4, 5, 6 & 8**	Exposed Sites **4, 5 & 6**	Exposed Sites **5, 6 & 8**
Permanent resident fauna	71	71	72	69

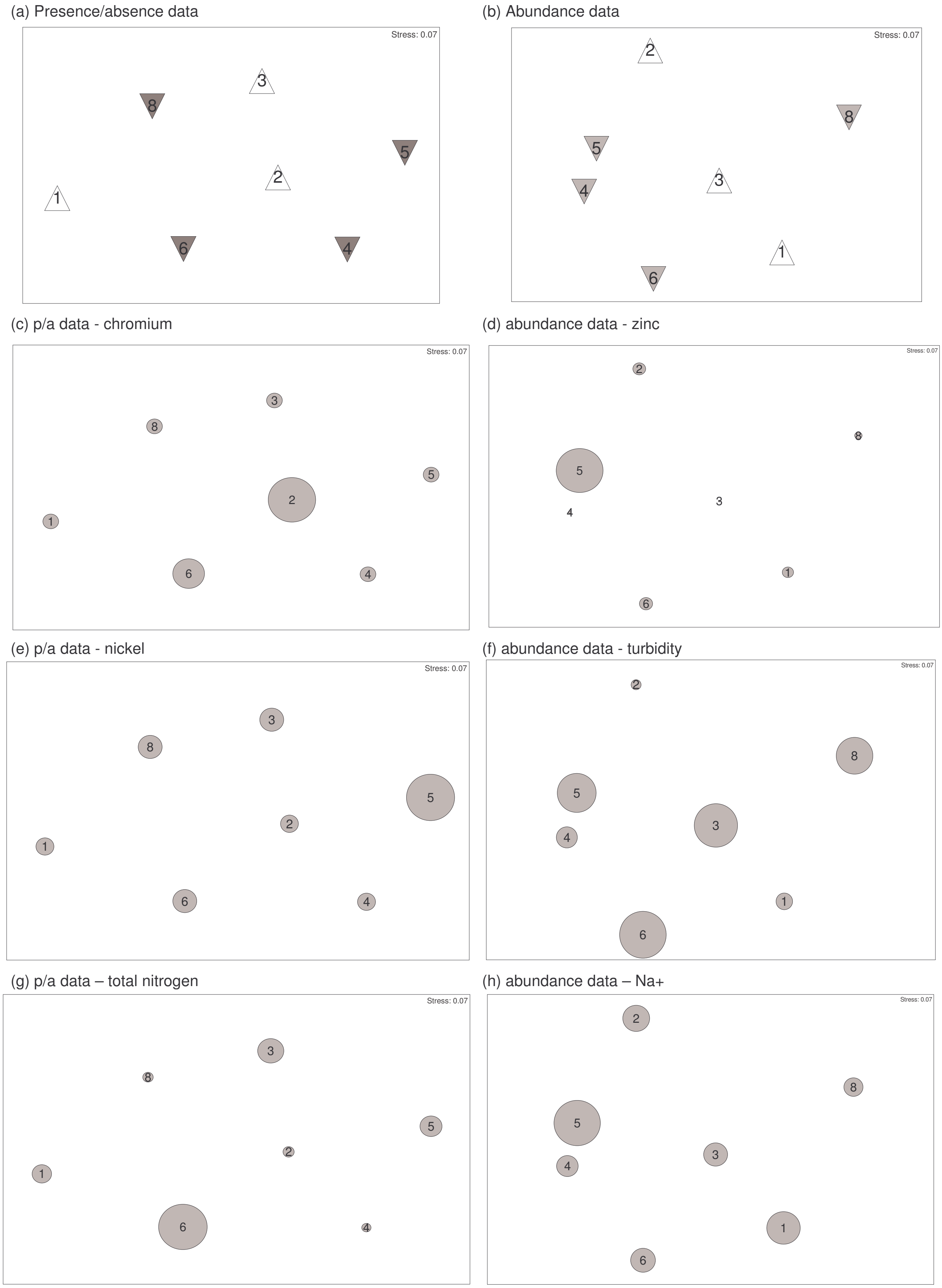


Figure 5. Two-dimensional MDS ordination of a) presence/absence data (stress 0.07) and b) abundance data (stress 0.07) for permanent invertebrate fauna from Jones Creek sampling sites with examples of likely influencing physico-chemical variables superimposed as 'bubbles' (c - h).
Codes for plots (a) and (b): ▼ exposed sites; △ control sites.

3.4 Aquatic Vertebrate Fauna

3.4.1 Fish

No fish species were recorded during the current study and none were recorded from the previous Streamtec (1992) survey. The episodic nature of flow in the creek and claypans and lack of permanent pools over the dry season mean it is highly unlikely that native fish could survive in the system. There may be permanent water bodies within the catchment, for example granite pools in the surrounding ranges, but these would need to be periodically connected to the stream in order for fish to reinvade each wet season. There is only one species of native fish, the spangled perch *Leiopotherapon unicolour*, which is known to be widespread throughout desert regions of Western Australia. However, there is no evidence that it can survive without permanent water, and its absence likely indicates that this system seldom (never) connects to permanent water with refuge populations.

3.4.2 Frogs

Tadpoles of the burrowing frog genus *Neobatrachus* sp. were present at all sites and were particularly abundant in Jones Creek. Tadpoles were too immature to identify to species-level with confidence, but were possibly *N. wilsmorei* (Goldfield's bullfrog), *N. suter* (Shoemaker frog) and/or *N. sudelli/centralis* (desert trilling frog) (M. Dziminski, UWA, pers. com.). All three *Neobatrachus* species are widely distributed through the mid western and mid-central parts of Western Australia and are often found in high densities in claypans after rain. Development is rapid with evidence that some species can accelerate development (to 17 days) if pools are particularly shallow, thus increasing their chances of survival (*e.g. N. centralis*; Read 1999). Two adult water holding frogs, *Cyclorana platycephala*, were also recorded from creek sites 4 and 6 (Plate 15). This species is one of a number of burrowing frog species common in arid zones across central Australia. Adults only appear above ground after rain to breed. Water holding frogs occur in a wide range of temporary waterbodies including creeks, pools and claypans (Main 1965, Cogger 1986). As with *Neobatrachus*, development and metamorphosis is rapid (about 30 days; Main 1965) and once waterbodies begin to dry, frogs make deep, water-tight chambers, which together with the frogs bladder is filled with water, enabling the frog to spend extended periods underground. Although longevity of Australian burrowing frogs is not known, it is suspected they may live for more than ten years (Read 1999).

Streamtec (1992) noted tadpoles of *C. platycephalus* to be abundant in Jones Creek in January 1992. They also recorded tadpoles of two other native burrowing frogs: *Limnodynastes spenceri* and *Neobatrachus* sp. (possibly *N. sutor* or *N. sudelli/centralis*). *L. spenceri* is common across arid regions of western and central Australia and although usually found in the sandy beds of temporary creeks, it can migrate into nearby ranges during higher rainfall periods to breed in rock holes (Cogger 1986).

(a)

(b)

Plate 15. Adult burrowing frogs *Cyclorana platycephalus*, (a) from sites 4 and (b) from site 6 on Jones Creek, May 2005.

3.5 Power Analysis to Determine Ability of Sampling Design to Detect Changes

Initially, one-way analysis of variance (ANOVA) was used to test for differences in species richness (total, permanent and temporary resident faunas) between upstream control sites (1, 2 & 3) and downstream exposed sites (4, 5 & 6). ANOVA detected a significant difference for temporary fauna (exposed > control), but with no significant effects for total fauna or permanent resident fauna (refer Table 6). This indicated that the current design had adequate power to detect the difference in species richness between control and exposed sites for temporary fauna. This also indicated a significant difference existed, whereas there was no difference for permanent or total fauna.

In the first instance, power analysis using the DESIGN module of SYSTAT was conducted to compare upstream control to downstream exposed sites for each faunal element. This was achieved by taking the observed mean species richness and progressively reducing the mean and assessing changes in replication needed to achieve adequate power. Analysis indicated that an excessive number of replicate samples (> 30) would be required to detect a 10% decline in species richness in permanent fauna between control and exposed sites, with the current 3 replicates able to detect a 40% effect size (Table 10). Eight replicates would be required to detect a 20% decline in species richness of permanent fauna.

Analysis of replicates required to detect a change at exposed sites over time indicated that 16 replicates would be required to detect a 10% decline in species richness in permanent fauna at exposed sites, with the current 3 replicates able to detect a 30% effect size (Table 10). Six replicates would be able to detect a 20% decline in species richness of resident fauna.

Table 12. Summary of power analysis on May 2005 Jones Creek invertebrate fauna data to determine level of replication required to detect specific effects sizes (% reductions) in species richness for total fauna, permanent fauna and temporary resident fauna between upstream control and downstream exposed sites.

Element	obs mean	10% effect size	20% effect size	30% effect size	40% effect size	50% effect size
Total fauna	51.0	18	6	3	2	2
Permanent fauna	32.3	34	8	5	3	3
Temporary fauna	18.7	13	5	3	3	2

Table 13. Summary of power analysis on May 2005 Jones Creek fauna data to determine level of replication required to detect specific effects sizes (% reductions) in species richness for total fauna, permanent fauna and temporary fauna at downstream exposed sites over time/sampling events.

Element	obs mean	10% effect size	20% effect size	30% effect size	40% effect size	50% effect size
Total fauna	59	7	3	3	2	2
Permanent fauna	36	16	6	3	3	2
Temporary fauna	23	10	4	3	3	2

It was considered that a 10% effect size (10% decline in aquatic invertebrate taxa richness) is a relatively stringent assessment criterion for detecting future impacts and would also require excessively high replication. However, analyses indicated that eight replicate samples from control and from exposed sites would allow the detection of a 20% decline in species richness and this was considered an acceptable assessment criterion. Any lesser level of replication would have a reduced ability to detect impacts, should they occur. It is therefore recommended that for ongoing monitoring of potential mine impacts to Jones Creek, at least eight replicate samples should be collected from upstream control and from downstream exposed reaches/sites.

4. CONCLUSIONS

4.1 Catchment Condition and Riparian Vegetation

Overall, Jones Creek and the south-west claypans area were considered extremely degraded due to historic disturbance of the riparian zones – loss of native vegetation and widespread channel and floodplain erosion. The greatest potential threats posed by nickel mining are associated with the alteration of streamflow regimes (through channel diversion, changes to sheet flow and reduction in extent of 'perched' water tables), further vegetation loss, accelerated erosion and increased soil and water contaminants. Any proposed management/monitoring programmes should seek to prevent further degradation of existing ecological values and to facilitate the restoration of any degraded areas. Further catchment degradation (*i.e.* loss of vegetative cover) will result in faster and greater run-off. As a result of greater flows and inputs into the claypans, with overflow to Lake Miranda to the south, there is the potential for head-cutting to develop in the south-west claypans that may eventually lead to their loss (H. Pringle, Ag. Dept., pers com.) and the loss of their unique faunal communities.

All remnant riparian vegetation, but in particular *Eucalyptus camaldulensis*, was considered to be a critically important component of the creek landscape. While the role of remnant riparian vegetation in intermittent arid-zone stream ecology is not well understood, it is likely to provide at least some of the energy, or 'the food source', that drives many aquatic processes. For example, leaf litter is likely to supply energy and nutrients essential to benthic and hyporheic organisms as well as providing shelter for invertebrates and tadpoles. The loss of deep-rooted riparian vegetation (trees and perennial shrubs) through grazing and past mining activities has already lead to considerable channel scour, bank erosion, sedimentation and a reduction in the general health of the ecosystem. Erosion in turn limits successful recruitment, as trees are usually restricted to stable soils. Increased sediment loads and highly mobile sand in-channel may result in increased pool aggradation and loss of surface bed complexity (Boulton *et al.* 2002), reducing the diversity of available in-stream habitat and hence reducing biodiversity. In addition, the high evaporation to rainfall ratio of the region means that loss of floodplain vegetation will also cause soil profiles to dry, water tables to drop and creeks to dry more rapidly.

While flood periodicity is known to be a major determinant of vegetation in arid regions (Friedel *et al.* 1993, Capon 2001), loss of creekline and floodplain trees does not necessarily always follow a reduction in frequency and duration of flooding. For example, *E. camaldulensis* in other parts of Australia can take up to 40% of their water requirements from creeks, but do not obtain all their water from drainage lines and ephemeral flooding, even when groundwater is highly saline (Thorburn *et al.* 1994). While trees can acclimate to changes in surface and groundwater levels, the rate of change must be gradual. Roots of this species are known to extend more than 10 metres below the surface, which would enable them to tap the full depth of wetted alluvial soils. Current knowledge of vegetation-hydrology relationships is limited, however studies in the Harding River catchment indicated river red gums close to the river were likely to have shallower rhizospheres (2 - 6 m depth) due to more frequent inundation from surface flow, with optimal requirements for flow once every 1 - 2 years and total or partial rhizosphere inundation for at least 1 - 3 months (R. Froend, Edith Cowan Uni., pers. comm.). Eucalypt vigour may provide a useful indicator of any detrimental changes to flow regime (*e.g.* changes to depth or duration of saturation of alluvial soils) within the Jones Creek riparian zone.

The current mine plan will result in the diversion of Jones Creek to the south-east and around the proposed pit, with the diversion re-entering the natural channel downstream of the pit. Pit construction and channel diversion will mean the inevitable loss of trees. Losses downstream of the diversion or extensive disturbance of fringing vegetation either by physical removal or by alteration of surface flow, must be limited. Stock access, trampling and grazing of the riparian zone will also significantly reduce the capacity for seedlings to recolonise. Very few juvenile trees were observed at any of the sites. Tree recruitment in arid zones is typically low with high seedling mortality rates and low seed bank reserves (Britton & Brock 1994, Brock *et al.* 2003). Successful recruitment requires specific climatic and hydrological events and may only occur naturally every 15 - 30 years (R. Froend, ECU, pers. comm.).

Given the projected size of the pit and the surrounding land topography, the diversion will require substantial excavation to develop a channel of adequate size, dimensions and elevation to act as a 'natural' creekline. Design features which should be considered include:

- heterogeneity in plan form (meanders as opposed to a straight channel) and longitudinal profile (pools and riffles),
- appropriate bank profiles to avoid a steep, incised channel,
- areas in the channel where silt/sand can deposit to avoid the diversion having a uniform rock substrate, providing 'habitat' for spores/eggs of invertebrate fauna to reside,
- construction of benches upon which riparian vegetation can ultimately establish (or be established using propagation and planting) and
- sufficient conveyance capacity to cater for low frequency but high magnitude rainfall/run off events.

The point of re-entry into the natural channel also needs careful design to avoid excessive velocities from the diversion into the natural creek. If channel diversion was to result in extremely high water velocities at the point of re-entry, this will increase energy and lead to bed and bank erosion and incision of the drainage line.

Along with loss of riparian vegetation, WRC (1997) highlighted siltation/sedimentation of pools as a threat to the ecology of rivers in arid and semi-arid areas. Sediment from erosion can smother benthic and hyporheic organisms, coat organic deposits and algae upon which the fauna depend as a food source, and then cover and in-fill habitat (*i.e.* covers woody debris, gravel beds, undercuts, hollows *etc*). As the silt deepens, the pools become shallower, water temperatures generally increase, as do diurnal changes in oxygen levels, making the pools less suitable for fauna. Eventually, the pools are totally in-filled and the fine sediments limit down-welling into hyporheic zones to the extent they no longer provide a refuge in the dry season. If this process occurs along the length of a creek, dramatic losses in biodiversity will eventuate. For example, species such as the fairy shrimp *Branchinella frondosa*, that currently occurs in very high number within Jones Creek does not appear tolerant of highly turbid waters and would be lost from the system.

With the existing highly mobile bed sediments of Jones Creek, there is the very real threat that, over time, any potential contaminants from mine activities (*e.g.* metals, hydrocarbons *etc*) will be transported far from their point of entry, adsorbing to suspended fine silts and clays to be finally deposited and concentrated in the south-west claypans. Under suitable conditions, these concentrated contaminants may be re-released and mobilised when next inundated. Continued monitoring of the claypans is therefore a critical component of the ongoing programme.

4.2 Water Quality and Metals

It is recommended that local water quality guideline values and indices be derived from background levels as measured in current baseline assessments. The ANZECC/ARMCANZ (2000) guidelines for general water quality parameters and for metals are conservative and as such do not necessarily reflect natural levels within the Jones Creek catchment. For example, Cr, Cu and Zn as well as Ni would appear to be 'naturally' elevated in catchment waters. Should elevated metals be detected at any stage, consideration should be given to the deployment of DGTs to determine bio-available components, with levels referred against ANZECC guidelines.

While fauna may have some level of tolerance/acclimation to naturally higher background levels, any further elevation due to mining could exceed thresholds of bio-available metals and result in lethal or sub-lethal toxic effects.

4.2 Aquatic Fauna

'Permanent' resident fauna (*i.e.* those with desiccation-resistant life stages), in particular micro-invertebrates and branchiopods, dominated both creek and claypan invertebrate communities. While most were cosmopolitan species, widespread throughout inland waters in arid regions of Western Australia, one new species of rotifer (*Cephalodella* sp. nov.) and range extensions (new to WA) for three other rotifers were recorded for the catchment. However, there have been few comprehensive studies of the taxonomy and distribution of microfauna in Western Australia, making it difficult to confer conservation status on these species.

Analyses showed no significant differences in assemblages or taxa richness of permanent fauna, with high between-site similarity in assemblage composition for control and exposed creek sites. This is a good baseline condition, with no current differences, allowing future differences to be detected. Temporary fauna was more variable in composition, and it is recommended that sampling does not target this component, but continues to assess the non-resident fauna as incidental fauna in samples taken to obtain resident fauna.

Permanent resident fauna will undergo a successional sequence of development with taxa likely appearing and disappearing over the seasonal water cycle (*i.e.* from the creek first flowing to pools receding and drying weeks to months later, depending on rainfall). Therefore, time of sampling after commencement of flow is important. The selective pressure on the permanent resident fauna is to emerge, develop, breed and produce resting eggs before temporary predators invade the system. Current sampling occurred approximately two weeks after flow commenced following late autumn/early winter rainfall. Life stages present were relatively diverse and sufficiently mature as to allow taxonomic identifications with confidence. Sampling earlier would make taxonomy difficult, while sampling later could mean fauna have developed and left the system or returned to resting stage. Therefore, sampling 2 - 3 weeks after the system commences to flow in autumn/winter is considered ideal.

Power analysis determined, that based on current inter-sample variability, an optimum replication of eight sites upstream and eight downstream will allow the detection of at least a 20% or greater change in taxa richness, should such a change occur. Sampling design should be modified in future monitoring to incorporate additional sites in the design.

The claypan site is likely to be the receiving environment for any potential contaminants that are washed down the system and will ultimately concentrate over time. The claypan supports a fauna distinct from that of the creek. Ideally replicate 'control' claypans should be identified and included in future monitoring (*i.e.* adjacent claypans that flood with a similar frequency, but do not directly receive run-off from Jones Creek. Currently there is one sample collected from the claypan. If additional claypans are not available, at a minimum, replicate samples should be collected from the claypan to allow detection of temporal changes in physico-chemical condition pre- and post mine development.

While the hyporheic zone is believed to form an important dry-season refuge for invertebrate fauna (Williams 1984, Smith *et al.* 2004), there have been few studies documenting the movement of fauna between the hyporheic zone and stream bed surface. Boulton (1989) theorised that in arid region streams with unstable and/or highly mobile sand substrates, the importance of the hyporheic zone as a refuge from drought and flood may be diminished due to high substrate temperatures and severe scouring by flash floods. DeLucchi (1989) also found that the hyporheos did not play a significant role as refuge in some intermittent streams in North America. Given the highly mobile bed substrate of Jones Creek and greater sampling effort required, it is therefore not recommended that hyporheic core or propagule bank (dry-phase) sampling be trialled at this stage. If eight replicate control and exposed sites are used to sample zooplankton and benthos as surveyed in the current study, this should be sufficient to detect any long-term changes in aquatic invertebrate fauna.

In addition to the invertebrate fauna, consideration should be given to the use of the currently abundant frogs in long-term bio-monitoring programmes for the Jones Creek catchment. Read (1999) noted that frogs may be of particular ecological importance in arid ecosystems, given their often high abundance and that they therefore may prove useful as bio-indicators of mining impacts. *Neobatrachus centralis* has previously been used as a bio-indicator of mining activities at WMC's Olympic Dam mine at Roxby Downs in South Australia (Read & Tyler 1994, Read 1997). Tadpoles are herbivores/detritivores and in Jones Creek may be dependent on algal food sources. As algae are known to 'sequester' certain metals there is the potential for bio-accumulation in tadpoles of *Neobatrachus* and *Cyclorana*, which could make them useful as bio-monitors.

Another method that may prove of value to long-term aquatic ecosystem monitoring within Jones Creek catchment, is the use of stable isotope (carbon & nitrogen) analysis to detect 'shifts' in food web structure. Given the (likely) low number of trophic levels in the early successional communities in the creek pools and claypan, stable isotope analysis should be relatively cost efficient due to the fewer samples required. It could be used to determine if the aquatic food-web is algal-driven, *i.e.* if algae play an important role supporting the food chain. Direct toxicity of metals (*e.g.* copper) often effects algae at much lower concentrations than effect higher organisms. If algae are important, then a decline in algal productivity would mean a reduction in availability of algal carbon and this would be detected (by stable isotope analysis) early on as a shift in the energy base of aquatic foodwebs towards (say) terrestrial (leaf detritus *etc*) carbon. If the system has no capacity to utilize terrestrial carbon, ecosystem collapse may ensue. Initial sampling would be required pre-mine to establish a baseline, however further analyses would not necessarily be required unless changes in invertebrate fauna and/or water quality were to be detected. It must be noted that while this approach has wide acceptance for general river health monitoring (*via* assessing trophic links), it has not been trialled previously for mine impact assessments in Australia.

5. RECOMMENDATIONS

5.1 Channel Diversion

Design of the diversion channel should consider and incorporate where practicable:

1. Conveyance capacity to cater for low frequency but high magnitude rainfall/run off events;
2. Replicating existing stream plan form with heterogeneity in bed profile, *i.e.* with the potential for meanders and depressions where streamflow can pool, as opposed to a straight channel;
3. Appropriate bank profiles to avoid a steep, incised channel;
4. Creation of alluvial bed habitats within the channel to act as dry-season refuges for invertebrate fauna;
5. In-channel benches for establishment of riparian vegetation;
6. Point of re-entry into the natural channel that avoids excessive velocities from the diversion into the natural creek;
7. Only limited vegetation losses (in particular mulga & eucalypts) downstream of the diversion and prevent extensive disturbance of fringing vegetation either by physical removal or by alteration of surface flow;
8. Stabilisation of banks using natural materials (*i.e.* woody debris, matting, rocks, brushing and revegetation);
9. Revegetation to limit erosion and sediment loading and reduce the run-off and excessive deposition of fine silts that may smother benthos and hyporheos;
10. Replanting/revegetation with local mulga and eucalypts adjacent to diversion channel to compensate loss of overstorey species;
11. Restriction of livestock access to rehabilitation zones;

5.2 Monitoring Aquatic Fauna

12. Sampling of all sites on, at least, a further two occasions prior to mining is recommended (depending upon adequate rainfall being received) to enhance the pre-disturbance baseline and help incorporate natural annual variation into the design;
13. Sampling design should be modified to incorporate an optimum eight replicate sites upstream (control sites) and eight downstream (exposed sites) to ensure detection of (possible) changes in species diversity related to mine activities. The eight sites downstream of the diversion will effectively form a gradient from east to west that will help pinpoint the source of any contaminant/effect. More sites can be added if greater resolution is required or sites can be re-positioned (*e.g.* along the eastern tributary) to target any likely known sources;
14. Replicate 'control' claypans should be identified and included in future monitoring. At a minimum, replicate samples should be collected from the 'exposed' claypan to allow detection of temporal changes in physico-chemical condition pre- and post mine development. In this instance, a 'gradient' approach should be used within the claypan for fauna and metal monitoring (*i.e.* establish a gradient of sites with distance away from the Jones Creek entry point);
15. Sampling should be conducted 2 - 3 weeks after the system commences to flow in autumn/winter;
16. Sampling should target 'permanent resident fauna, in particular micro-invertebrates (protista, rotifers, ostracods & copepods) and branchiopods (water fleas, clam shrimp, shield shrimp & fairy shrimp), but allow for inclusion of any 'temporary' fauna present;

17. For comparative purposes, future sampling should adopt methods consistent with those used in the current baseline study;
18. Investigate the potential of burrowing frogs (*Neobatrachus* spp. & *Cyclorana platycephalus*) as bio-indicators of mine impacts; tadpoles for bio-accumulation studies could be collected concurrent with invertebrate sampling 3 weeks after the system commences to flow;
19. Investigate the potential of isotope analysis (carbon & nitrogen) as a relatively cost efficient 'early warning' indicator of ecosystem change (as per ANZECC/ARMCANZ 2000 recommended ecosystem function approaches to monitoring). Consider an initial single trial as part of a risk analysis approach to potential metal pollutants, with the aim of developing a baseline. If any changes in fauna or water quality were to be detected at some later date, stable isotope analyses could then be used to determine the extent of the impact; *i.e.* whether or not overall ecosystem function/health was also affected.
20. Once constructed, bio-monitoring sites should be established within the diversion channel to determine effectiveness of the channel design in approximating habitat originally provided by the natural creek channel;
21. Long-term monitoring of the aquatic ecosystem should continue for the life of the mine.

5.3 Other

22. Include in long term monitoring, the monitoring of eucalypt vigour as an indicator of change in flow regime.

REFERENCES

ANZECC/ARMCANZ (2000a). Australian and New Zealand Guidelines for Fresh and Marine Water Quality. Australia and New Zealand Environment and Conservation Council and the Agriculture and Resource Management Council of Australia and New Zealand. Paper No. 4. Canberra. http://www.deh.gov.au/water/quality/nwqms/index.html

ANZECC/ARMCANZ (2000b). Australian and New Zealand Guidelines for Water Quality Monitoring and Reporting. Australia and New Zealand Environment and Conservation Council and the Agriculture and Resource Management Council of Australia and New Zealand. Paper No. 7. Canberra. http://www.deh.gov.au/water/quality/nwqms/index.html

Bayly I.A.E & Williams W.D. (1973). "Inland Waters and their Ecology". Longman, Melbourne.

Bunn S.E. (1988). Processing of leaf litter in a northern jarrah forest stream, Western Australia: seasonal differences. *Hydrobiologia* **162**: 201- 210.

Bunn S.E., Edward D.H. & Loneragan N.R. (1986). Spatial and temporal variation in the macroinvertebrate fauna of streams of the northern jarrah forest, Western Australia: community structure. *Freshwater Biology* **16**: 67-91.

Bunn S.E., Davies P.M. & Edward D.H.D. (1989). The association of *Glacidorbis occidentalis* Bunn & Stoddart (Gastropoda: ?Hydrobiidae) with intermittently- flowing forest streams in south-western Australia. *Journal of the Malacology Society of Australia* **10**: 25-34.

Boulton A.J. (1989). Over-summering refuges of aquatic macroinvertebrates in two intermittent streams in central Victoria. *Transactions of the Royal Society of South Australia* **113**: 23-34.

Boulton A.J. (2002). The functional role of the hyporheic zone. *Verhandlungen Internationale Vereinigung Limnologie* **27**: 51-63.

Boulton A.J., Depauw S. & Marmonier P. (2002). Hyporheic dynamics in a degraded rural stream carrying a 'sand slug'. *Verhandlungen Internationale Vereinigung Limnologie* **28**: 120-124.

Boulton A.J. & Lake P.S. (1988). Australian temporary streams: some ecological characteristics. *Verhandlungen Internationale Vereinigung Limnologie* **23**:1380-1383.

Boulton A.J. & Suter P.J. (1986). Ecology of Temporary Streams – an Australian Perspective. [In] P. De Deckker & W.D. Williams (eds) "Limnology in Australia". CSIRO, Australia and Dr W. Junk, Dordrecht, The Netherlands. Pp. 313-327.

Borenstein M., Rothstein H. & Cohen J. (1997). "Power and Precision." Biostat Inc., Teaneck, New Jersey.

Britton D.L. & Brock M.A. (1994). Seasonal germination from wetland seed banks. *Australian Journal of Marine and Freshwater Research* **45**: 1445-1457.

Brock M.A., Nielson D.L., Shiel R.J., Green J.D. & Langley J.D. (2003). Drought and aquatic community resilience: the role of eggs and seeds in sediments of temporary wetlands. *Freshwater Biology* 48 (7): 1207-1218.

Burnside D., Holm A., Payne A. & Wilson G. (1995). "Reading the Rangeland: A Guide to the Arid Shrublands of Western Australia". Department of Agriculture, Western Australia, Perth.

Capon S.J. (2001). The Effects of Flood Frequency on Floodplain Vegetation in a Variable and Arid Catchment; Cooper Creek, south-west Queensland. [In] Review for Proceedings of the Royal Society of Queensland Lanscape Health, 2000.

Claret C. & Boultin A.J. (2003). Diel variation in surface and subsurface microbial activity along a gradient of drying in an Australian sand-bed stream. *Freshwater Biology* **48**: 1739-1755.

Clarke K.R. & Gorley R.N. (2001). Primer v5: User Manual/Tutorial, Primer E: Plymouth. Plymouth Marine Laboratory, Plymouth, UK.

Clarke K.R. & Warwick R.M. (2001). "Changes in Marine Communities: An Approach to Statistical Analaysis and Interpretation". 2nd edition. Primer E: Plymouth. Plymouth Marine Laoratory, Plymouth, UK.

Clifford H.F. (1966). The ecology of invertebrates in an intermittent stream. *Investigations of Indiana Lakes & Streams* **7**(2): 57-98.

Cogger H.G. (1986). "Reptiles and Amphibians of Australia". A.H. & A.W. Reed, Sydney.

Cohen, J. (1988) Statistical Power Analysis for the Behavioural Sciences. 2nd edition, Hillsdale, New Jersey, USA.

Davies P.M. & Bunn S.E. (2003). Bio-monitoring Temporary Streams: Issues and Methodologies. [In] R. McLean (ed) Proceedings of a Workshop on Water Quality Issues in Final Voids, Salt Lakes and Ephemeral Streams, Perth. ACMER and Centre for Sustainable Mine Lakes, Brisbane.

Davis J.A., Harrington S.A. & Friend J.A. (1993). Invertebrate communities of relict streams in the arid zone; the George Gill Range, central Australia. *Australian Journal of Marine and Freshwater Research* **44**: 483-505.

DeLucchi C.M. & Peckarsky B.L. (1989). Life history patterns of insects in an intermittent and permanent stream. *Journal of the North American Benthological Society* **8**(4): 308-321.

Dept. of Ag. (1992). "Rangeland Management in Western Australia". Department of Agriculture, Western Australia, Miscellaneous Publication 8/92. May 1992.

ecologia (1990). Yakabindie Nickel Mine Project Consultative Environmental Review: Flora and Fauna Survey. Unpublished report to Dominion Mining Limited by Ecologia Ecological Consultants. March 1990.

Edward D.H., Gazey P & Davies P.M. (1994). Invertebrate community structure related to physico-chemical parameters of permanent lakes of the south coast of Western Australia. *Journal of the Royal Society of Western Australia* **77**: 51-63

Fairweather P.G. (1991). Statistical power and design requirements for environmental monitoring. *Australian Journal of Marine and Freshwater Research* **42**: 555-567.

Friedel M.H., Pickup G. & Nelson D.J. (1993). The Interpretation of vegetation change in a spatially and temporally diverse arid Australian landscape. *Journal of Arid Environments* **24** (3):241-260.

Hairston N.G. Jr, Van Brunt R.A., Kearns C.M. & Engstrom D.R.. (1995). Age and survivorship of diapausing eggs in a sediment egg bank. *Ecology* **76**: 1706-1711.

Hall N.J., McKenzie N.L. & Keighery G.J. (1994). The biological survey of the eastern goldfields of Western Australia. Part 10, Sandstone-Sir Samuel and Laverton-Leonora study area. *Records of the Western Australian Museum Supplement* **47**: 1-166.

Halse S.A. (2002). Diversity of Ostracoda (Crustacea) in inland waters of Western Australia. *Verhandlungen Internationale Vereinigung Limnologie* **28**: 914-918.

Halse S.A. & Storey A.W. (1996). Aquatic invertebrate surveys and water quality of Perth Airport swamps. Unpublished report to the Perth Airport Authority, 23 pp.

Halse S.A., Pearson G.B. & Kay W.R. (1998). Arid-zone networks in time and space: waterbird use of Lake Gregory in north-western Australia. *International Journal of Ecology and Environmental Sciences* **24**: 207-222

Hart B.T. & McKelvie I.D. (1986). Chemical Limnology in Australia. [In] P. De Deckker & W.D. Williams (eds) "Limnology in Australia". CSIRO/Dr Junk Publishers. Pp 3 - 32.

IUCN (2004). 2004 Red List of Threatened Species. International Union for Conservation of Nature and Natural Resources. http://www.redlist.org/

Jenkins K.M., Boulton A.J. & Gawne B. (2002). Colonisation pathways of microinvertebrates following flooding in a dryland Australian river. *Verhandlungen Internationale Vereinigung Limnologie* **28**: 1444-1447.

John J., Dunlop N., Goldberg J., Lowe E. & Osbourne J. (2000). Baseline Aquatic Biology Survey of Lake Miranda, 26 September 2000, Department of Environmental Biology, Curtin University of Technology, Perth.

John J., Lowe E. & Butson F. (2003). Lake Miranda Study. Department of Environmental Biology, Curtin University of Technology, Perth.

Johnson S.L., Commander D.P. & O'Boy C.A. (1999). Groundwater Resources of the Northern Goldfields, Western Australia: Water and Rivers Commission, Hydrogeological Record Series, Report HG 2, 57p.

Lake P.S., Barmuta L.A., Boulton A.J., Campbell I.C. & St Clair R.M. (1985). Australia streams and Northern Hemisphere stream ecology: comparisons and problems. *Proceedings of the Ecological Society of Australia* **14**: 61-82.

Lovett S. & Price P. (1999). Riparian Land Management Technical Guidelines, Volume One. Principals of Sound Management. Land & Water Resources Research & Development Council. Canberra.

Kennedy J.H., Cook R.E. & Anderson G.M. (1998). Diversity and Temporal Patterns of Macroinvertebrates Inhabiting Southern High Plains Playas. [In] A.J. McComb & J.A Davis (eds) "Wetlands for the Future". Gleneagles Publishing. Pp. 121-137.

Main A.R. (1965). Frogs of southern Australia. Handbook No. 8. Western Australia Naturalist Club, Perth.

Metzeling L., Robinson D., Perriss S. & Marchant R. (2002). Temporal persistence of benthic invertebrate communities in south-eastern Australian streams: Taxonomic resolution and implications for the use of predictive models. *Marine and Freshwater Research* **53**: 1223-1234.

Mitchell A.A. & Wilcox D.G. (1994). "Arid Shrublands of Western Australia". University of Western Australia Press in association with the Department of Agriculture, Western Australia.

McKeon G.M., Hall W., Henry B.H., Stone G. & Watson I.W. (Eds) (2004). Pasture Degradation and Recovery in Australia's Rangelands. Learning from History. Queensland Department of Natural Resources, Mines and Energy. Brisbane, Australia. 256 pp.

Morton S.R., Short J. & Barker R.D. (2004). Refugia for Biological Diversity Australia. Biodiversity Series Paper 4. Retrieved: February. http://www.deh.gov.au/biodiversity/publications/series/paper4/bio1023.html

OECD (1982) Eutrophication of waters: monitoring, assessment and control. Organisation for Economic Coop-eration and Development, Paris.

Paltridge R.M., Dostine P.L, Humphrey C.L & Boulton A.J. (1997). Macroinvertebrate recolonization after re-wetting of a tropical seasonally-flowing stream (Magela Creek, Northern Territory, Australia) *Marine and Freshwater Research* **48**: 633–645.

Pen L. & Scott M. (1995). Stream Foreshore Assessment in Farming Areas. Blackwood Catchment Co-ordinating Group. National Landcare Program. Department of Agriculture WA. June 1995.

Pontin R.M. & Shiel R.J. (1995). Periphytic rotifer communities of an Australian seasonal floodplain pool. *Hydrobiologia* **313/314**: 63–67.

Porter J., Brock M.A & Kingsford R. (2001). Seed bank dynamics in an arid zone saline wetland. Ecological Society of Australia, Annual Conference, University of New England, Armidale, NSW, 8-10 December, p. 126.

Pringle H. & Tinley K. (2003). Are we overlooking critical geomorphic determinants of landscape change in Australian rangelands? *Ecological Management and Restoration* **4**: 180-186.

Pringle H.J., Watson I.W. & Tinley K.L. (in press). Rangeland improvement, or ongoing degradation – reconciling apparent contradictions from the arid rangelands of Western Australia. *Landscape Ecology*.

Rangelands NRM Co-ordinating Group (2005a). Draft Gascoyne-Murchison Natural Resource Management Plan. Rangelands NRM Co-ordinating Group and the Interrim Gascoyne Murchison natural Resource Management Council. January 2005. http://www.rangelandswa.info/downloads/gas-mur_draft.pdf

Rangelands NRM Co-ordinating Group (2005b). Draft Natural Resource Management Strategy for the Rangelands Region of Western Australia. April 2005.

Read J.L. (1997). Comparative abnormality rates of the trilling frog at Olympic Dam. *Herpetofauna* **27**: 23-27.

Read J.L. (1999). Abundance and recruitment patterns of the trilling frog (*Neobatrachus centralis*) in the Australian arid zone. *Australian Journal of Zoology* **47**: 393-404.

Read J.L. & Tyler M.J. (1994). Natural levels of abnormalities in the trilling frog (*Neobatrachus centralis*) at the Olympic Dam mine. *Bulletin of Environmental Contamination and Toxicology* **53**: 25-31.

Salas H.J. & Martino P. (1991). A simplified phosphorus trophic state model for warm-water tropical lakes. *Water Research* **25**: 341-350.

Sheppard C.R.C. (1999). How large should my sample be? Some quick guides to sample size and the power of tests. *Marine Pollution Bulletin* **38**:439-447.

Sheldon F., Boulton A.J. & Puckridge J.T. (2002). Conservation value of variable connectivity: aquatic invertebrate assemblages of channel and floodplain habitats of a central Australian arid-zone river, Cooper Creek. *Biological Conservation* **103**: 13 31.

Shiel R.J., Green J.D. & Tan L.W. (2002). Microfaunal and resting-stage heterogeneity in ephemeral pools, upper River Murray floodplain, Australia. *Verhandlungen Internationale Vereinigung Limnologie* **27**: 3738-3741.

SKM (2005). Jones Creek Stream Sediment Characterisation. Unpublished draft report by Sinclair Knight Merz Pty Ltd to WMC Resources Limited. May 2005.

Smith M.J. & Storey A.W. (2001). Design and Implementation of Baseline Monitoring (DIBM3): Developing and Ecosystem Health Monitoring Program for Rivers and Streams in Southeast Queensland. Report to the South East Queensland Regional Water Quality Management Strategy, Brisbane, 416p.

Smith R., Jeffree R., John J. & Clayton P. (2004). Review of Methods for Water Quality Assessment of Temporary Stream and Lake Systems. Unpublished report to the Australian Centre for Mining Environmental Research, Kenmore, Qld. September 2004.

Storey A.W., Halse S.A. & Shiel R.J. (1993). Aquatic invertebrate fauna of the Two Peoples Bay area, southwestern Australia. *Journal of the Royal Society of Western Australia* **76**:25-32.

Storey A.W., Shiel R.J. & Lynas J. (2004). Buntine-Marchagee Natural Diversity Recovery Catchment (BMRC) Wetland Invertebrate Fauna Monitoring: August 2004. Unpublished report by the School of Animal Biology, The University of Western Australia to the Department of Conservation and Land Management, Mid-West Regional Office. pp 59.

Streamtec (1992). Yakabindie Nickel Mine Project: Jones Creek Baseline Survey of Aquatic Fauna and Water Quality. Unpublished report by Streamtec Pty Ltd Ecological Consultants to Dominion Mining Limited. Report ST 140. February 1992.

Timms B.V. (2002). Limnology of the claypans of the Paroo, arid-zone Australia. *Verhandlungen Internationale Vereinigung Limnologie* **28**: 130-133.

Timms B.V. & Boulton A.J. (2001). Typology of arid-zone floodplain wetlands of the Paroo River, inland Australia and the influence of water regime, turbidity, and salinity on their aquatic invertebrate assemblages. *Archiv fur Hydrobiologie* **153**: 1-27.

Timms B.V., Datson B. & Coleman M. (IN PREP.). The Wetlands of the Lake Carey catchment, northeast Goldfields of Western Australia, with special reference to large branchiopods.

Thorburn P.J., Mensforth L.J. & Walker G.R. (1994). Reliance of creek-side river red gums on creek water. *Australian Journal of Marine and Freshwater Research* **45**: 1439-1443.

Towns D.R. (1985). Limnological characteristics of a South Australian intermittent stream, Brown Hill Creek. *Australian Journal of Marine and Freshwater Research* **36**: 821-837.

van Beurden E. (1984). Survival strategies of the Australian water-holding frog, *Cyclorana platycephalus*. [In] H.G. Cogger & E.E. Cameron (eds) "Arid Australia". Australian Museum, Sydney. pp. 223-234.

Wetzel R.G. (1983). "Limnology". 2nd Edition, Saunders College Publishing.

Wilcox D. (1960). The grazing of Wanderrie grass associations. *Journal of Agriculture Western Australia*. Bulletin **2759**.

Williams D.D. (1984). The hyporheic zone as a habitat for aquatic insects and associated arthropods. [In] V.H. Resh & D.M. Rosenberg (eds) "The Ecology of Aquatic Insects". Praeger, New York. Pp. 430–50.

Williams D.D. & Hynes H.B.N. (1977a). The ecology of temporary streams: I. The faunas of two Canadian streams. *Internationale Revue der gesamten Hydrobiologie* **61**(6): 761-787.

Williams D.D. & Hynes H.B.N. (1977b). The ecology of temporary streams: II. General remarks on temporary streams. *Internationale Revue der gesamten Hydrobiologie* **62**(1): 53-61.

Williams W.D. (1980). "Australian Freshwater Life, the Invertebrates of Australian Inland Waters". McMillan Co., Autralia.

Williams W.D. (1998). Diversity and Evolution of the Fauna of Dryland Wetlands. [In] A.J. McComb & J.A Davis (eds) "Wetlands for the Future". Gleneagles Publishing. Pp. 167-172.

Williams W.D. (2002). Biodiversity in temporary wetlands of dryland regions. *Verhandlungen Internationale Vereinigung Limnologie* **27**: 141-144.

Williams M. & Choo M. (1994). Environmental Monitoring Issues. Unpublished draft report to the Department of Conservation & Land Management, Western Australian. 10 pp.

WRC (1997). The State of the Northern Rivers. Publication No. WRAP 10, published by the Water & Rivers Commission, Perth, Western Australia.

WRC (1999). Planning and Management: Foreshore Condition Assessment in Urban and Semi-rural Areas of South-west Western Australia. Water and Rivers Commission River Restoration Report No. RR2. ISBN 0-7309-7443-X. October 1999.

APPENDICES

Appendix 1. Summary of riparian condition and physico-chemical and data from each site.

Site No.	Date	Weather Condition	GPS (WGS84) south	GPS (WGS84) east	Site Description	Riparian Condition
1	20/05/2005	50% cloud cover, blue skies.	27º25'02"	120º35'21"	Jones Creek Northern Tributary. Most upstream control site. Fairly confined channel approx. 12 m wide x 2 m deep; main pool 40 x 4 x 0.25m; flood debris > bankfull; some bank erosion; coarse sand/gravel bed - mobile sand waves approx. 30 cm high; pools along edges and around tree roots; numerous smaller pools; waters tannin stained - with leaf packs; lots of anostracans, conchostracans & tadpoles.	Fair-poor; degraded
2	18/05/2005	High broken cloud.	27º25'30"	120º34'54"	Jones Creek Northern Tributary. Within future pit area and just inside western boundary of Wanjarri NR. Tributary (anabranch??) from south-east series of smaller 15 x 3 x 0.3m over 200m of creek; channel confined, 2m high banks x 12m wide; flow debris indicates velocity of 30 m^3/sec @ 1 m/sec or 45 m^3/sec @ 1.5 m/sec; sand/gravel/pebble bed; banks to 50 cm; water tannin stained from leaves, otherwise clear. Extensive bank slumping & down-cutting.	Poor; extremely degraded
3	18/05/2005	High broken cloud.	27º25'52"	120º34'43"	Jones Creek Northern Tributary. Within future pit area. 100 m upstream of Six Mile Well pit crossing; series of smaller pools continue several hundred metres upstream 30 x 8 x 40m; deepest ~1.5m; coarse sand and gravel substrate, mobile bed; tannin stained.	Fair-poor; degraded
4	19/05/2005	100% cloud cover, light rain.	27º26'58"	120º34'05"	Jones Creek Northern Tributary. Numerous small - medium pools, generally interspersed amongst sand/gravel bars; coarse to medium, sand and gravel/pebbles; main pool 50 x 8 x 0.4 m; lot of erosion along banks, mobile sand, braided channel; lot of tadpoles; single adult *Cyclorana platycephalus* observed.	Fair-poor; degraded
5	18/05/2005	High broken cloud. Easterly breeze, wave action may aerate water.	27º27'59"	120º33'35"	Jones Creek main channel at Old Hwy Crossing. Main pool ~200 x 15 x 0.75m; numerous small, shallow pools along bank edge; coarse sand and gravel substrates, mobile unconsolidated bed, sand waves to around 30 cm deep; braided channel downstream; lot of tadpoles; adult odonates observed.	Fair-poor; degraded
6	19/05/2005	100% cloud cover.	27º29'39"	120º30'42"	Jones Creek main channel west of Goldfields Hwy. Creek crossing under powerline; banks badly eroded down to coffee rock - terracing; sediment finer than upstream, some pebbles but mostly silt and finer sands, unconsolidated – 'quick sand'; coffee rock exposed in terraces; small and large pools 200 x 10 x 0.3 m; smaller pools along bank edge or around tree roots (scour points); adult *Cyclorana platycephalus* observed.	Poor, extremely degraded
7	19/05/2005	100% cloud cover, light rain.	27º31'52"	120º26'57"	Northern-most of the SW Claypans, west of Goldfields Hwy. Very large, irregular claypan, flooded into fringing vegetaion; water very turbid with fine suspended clay; water level receded approx. 30 cm from full level (clay water mark on vegetation); depth gauge installed in 1.36 m depth of water.	Fair-poor; degraded
8	20/05/2005	50% cloud cover, blue skies.	27º27'53"	120º33'48"	Jones Creek Eastern Tributary. one main pool (30 x 8 x 0.3) and smaller pools up and downstream; channel very eroded; coffee rock exposed in terraces.	Poor, extremely degraded

Appendix 1. (cont.)

General water quality parameters and nutrient concentrations

Site No.	Date	Time (hrs)	Secchi (m)	Turbid (NTU)	Temp (ºC)	DO (%)	DO (mg/L)	pH	NO_3 (mg/L)	TN (mg/L)	P-SR (mg/L)	TP (mg/L)	Max. depth (m)
1	20/05/2005	1300	> depth	1.5	19.2	88	7.5	6.60	0.42	0.97	<0.01	0.06	0.25
2	18/05/2005	1500	> depth	0.9	21.8	105	8.5	6.28	0.40	0.56	<0.01	0.03	0.3
3	18/05/2005	1200	> depth	3.9	22.3	104	8.1	6.20	0.76	1.30	<0.01	0.02	1.5
4	19/05/2005	1550	> depth	1.9	20.4	114	9.5	6.74	0.13	0.46	<0.01	0.02	0.4
5	18/05/2005	0900	> depth	3.5	19.4	97	8.5	5.12	0.75	1.10	<0.01	0.03	1.5
6	19/05/2005	1300	> depth	4.2	20.3	106	8.9	6.57	1.90	2.40	<0.01	0.02	0.3
7	19/05/2005	0900	0.06	320	19.6	74	6.5	6.49	0.25	0.94	0.01	0.05	1.36 at gauge
8	20/05/2005	0900	> depth	3.3	18.5	114	9.9	6.83	0.06	0.52	<0.01	0.02	0.3

Electrical conductivity and composition of major ions

Site No.	Date	Time (hrs)	Cond. (µS/cm)	Ca (mg/L)	Na (mg/L)	K (mg/L)	Mg (mg/L)	HCO_3 (mg/L)	Cl (mg/L)	SO_4 (mg/L)
1	20/05/2005	1300	123.9	7.5	8.8	3.6	3.3	37	17	4.9
2	18/05/2005	1500	132.1	10.8	7.1	3.7	4.0	49	12	3.7
3	18/05/2005	1200	114.8	8.4	6.3	3.8	3.1	34	12	3.8
4	19/05/2005	1550	94.3	6.5	5.7	3.0	2.6	34	11	3.9
5	18/05/2005	0900	101.5	6.3	12.2	3.2	2.8	31	11	4.6
6	19/05/2005	1300	121.9	9.4	6.5	3.6	3.6	37	10	4.3
7	19/05/2005	0900	52.0	1.7	3.5	6.2	1.0	15	<5	0.7
8	20/05/2005	0900	79.7	4.3	5.0	4.1	2.7	34	8	2.0

Metal concentrations

Site No.	Date	Time (hrs)	As (mg/L)	Cd (mg/L)	Cr (total) (mg/L)	Cu (mg/L)	Hg (mg/L)	Ni (mg/L)	Pb (mg/L)	Se (mg/L)	Zn (mg/L)
1	20/05/2005	1300	<0.001	<0.0001	<0.002	0.003	Not analysed	0.003	0.0001	<0.001	0.029
2	18/05/2005	1500	<0.001	<0.0001	0.003	0.003	<0.0005	0.003	0.0001	<0.001	0.033
3	18/05/2005	1200	<0.001	<0.0001	<0.002	0.003	<0.0005	0.004	0.0002	<0.001	0.005
4	19/05/2005	1550	<0.001	<0.0001	<0.002	0.003	<0.0005	0.003	<0.0001	<0.001	0.007
5	18/05/2005	0900	<0.001	<0.0001	<0.002	0.007	<0.0005	0.008	0.0020	<0.001	0.120
6	19/05/2005	1300	<0.001	<0.0001	0.002	0.003	<0.0005	0.004	0.0001	<0.001	0.034
7	19/05/2005	0900	0.001	<0.0001	0.017	0.005	<0.0005	0.010	0.0010	<0.001	0.018
8	20/05/2005	0900	<0.001	<0.0001	<0.002	0.004	<0.0005	0.004	<0.0001	<0.001	0.019

Appendix 2. ANZECC/ARMCANZ (2002a) guideline levels for freshwaters.

(a) General Water Quality Parameters

NB: these trigger values are based on northern Qld, NT and regions north of Carnarvon in WA. There was no data available for northern WA rivers and no data specific to goldfields region when these trigger values were formulated.

Table A2-1. Trigger values for nutrients, dissolved oxygen and pH applicable to north-west Western Australia (TP = total phosphorus; FRP = filterable reactive phosphorus; TN = total nitrogen; NOx = total nitrates/nitrites; NH4+ = ammonium).

Ecosystem Type	TP (mg/L)	FRP (mg/L)	TN (mg/L)	NOx (mg/L)	NH4+ (mg/L)	DO % saturation[2]		pH	
						Lower	Upper	Lower	Upper
Upland River[1]	0.01	0.005	0.15	0.03	0.006	90	120	6.0	7.5
Lowland River[1]	0.01	0.004	0.2-0.3	0.01	0.01	85	120	6.0	8.0
Lakes & Reservoirs	0.01	0.005	0.35[3]	0.01	0.01	90	120	6.0	8.0
Wetlands[4]	0.01-0.05	0.005-0.025	0.35-1.2	0.01	0.01	90	120	6.0	8.0

Na = not applicable.
[1] All values during base river flow not storm events.
[2] Derived from daytime measurements; may vary diurnally and with depth; data loggers required to assess variability.
[3] Represents turbid lakes only.
[4] Higher nutrient values indicative of northern WA pools.

Table A2-2. Trigger values for conductivity and turbidity applicable to north-west Western Australia.

Ecosystem Type	Cond. (µS/cm)	Explanatory notes
Upland & lowland rivers	20 - 250	Values will vary depending on geology. First flush after seasonal rain may result in temporarily high values.
Lakes, reservoirs & wetlands	90 - 900	Higher values will occur during summer when water levels are reduced due to evaporation. Values in WA wetlands may exceed 900 µS/cm.
	Turbid. (NTU)	
Upland & lowland rivers	2 - 15	
Lakes, reservoirs & wetlands	2 - 200	Shallow lakes and reservoirs may have higher turbidity naturally due to wind-induced re-suspension of sediments. Lakes & reservoirs in catchments with highly dispersible soils will have high turbidity. Wetlands vary greatly in turbidity depending on general catchment condition, recent flow events and water level in the wetland.

(b) Metals

Table A2-3. Trigger values for metals for the protection of aquatic ecosystems. Values for slightly to moderately disturbed systems are shaded grey.

Metal		Trigger Value for Freshwater (mg/L)			
		Level of protection (% species)			
		99	95	90	80
Aluminium	pH >6.5	0.027	0.055	0.080	0.150
Aluminium	pH <6.5	ID	ID	ID	ID
Arsenic (III)		0.001	0.024	0.094[C]	0.360[C]
Arsenic (V)		0.0008	0.013	0.042	0.140[C]
Cadmium	[H]	0.00006	0.0002	0.0004	0.0008
Chromium (III)	[H]	ID	ID	ID	ID
Chromium (VI)		0.00001	0.001[C]	0.006	0.040
Copper	[H]	0.001	0.0014	0.0018[C]	0.0025[C]
Lead		0.0010	0.0034	0.0056	0.0094[C]
Mercury (inorganic)	[B]	0.00006	0.0006	0.0019[C]	0.0054[C]
Nickel		0.008	0.011	13	0.017
Selenium	[B]	0.005	0.011	0.018	0.034
Zinc	[H]	0.0024	0.008[C]	0.015[C]	0.031[C]

ID = indeterminate;
[B] Metals for which bioaccumulation and secondary poisoning effects should be considered;
[C] Level may not protect key test species from chronic toxicity;
[H] Metals for which levels should be adjusted if water hardness (as CaCO3) >30mg/L - refer Tables A2-4 and A2-5 below.

Table A2-4. Approximate factors to apply to trigger values (TV) in Table A2-3 for freshwaters of varying hardness. If water hardness is away from the mid-range, it may be preferable to use the algorithm in Table A2-5.

Hardness category (mg $CaCO_3$/L)	Mid-Range Hardness (mg $CaCO_3$/L)	Cd	Cr(III)	Cu	Pb	Ni	Zn
Soft (0-59)	30	TV	TV	TV	TV	TV	TV
Moderate (60-119)	90	x 2.7	x 2.5	x 2.5	x 4.0	x 2.5	x 2.5
Hard (120-179)	150	x 4.2	x 3.7	x 3.9	x 7.6	x 3.9	x 3.9
Very Hard (180-240)	210	x 5.7	x 4.9	x 5.2	x 11.8	x 5.2	x 5.2
Extremely Hard (400)	400	x 10.0	x 8.4	x 9.0	x 26.7	x 9.0	x 9.0

Table A2-5. General hardness algorithms describing guideline values, where: HMTV = hardness modified trigger value; TV = trigger value; H = measured hardness of the water body.

Metal	Hardness-dependent algorithm
Cd	$HMTV = TV\ (H/30)^{0.89}$
Cr (III)	$HMTV = TV\ (H/30)^{0.82}$
Cu	$HMTV = TV\ (H/30)^{0.85}$
Pb	$HMTV = TV\ (H/30)^{1.27}$
Ni	$HMTV = TV\ (H/30)^{0.85}$
Zn	$HMTV = TV\ (H/30)^{0.85}$

Appendix 3. Invertebrate taxa occurrences across sites.

Occurrence of each taxon across the eight sites sampled in the Jones Creek catchment, indicating fauna likely to be permanent residents (P) and those likely to be temporary residents (T). Abundances are $\log_{10}$ scale classes: 1 = 1 individual, 2 = 1- 10, 3 = 11 – 100, 4 = 101 - 1000, 5 = >1000.

TAXA			SITE							
		P/T	*1*	*2*	*3*	*4*	*5*	*6*	*7*	*8*
PROTISTA										
CILIOPHORA										
	cf. *Epistylis*	**P**	0	0	0	1	0	0	0	0
RHIZOPODA										
ARCELLIDAE										
	Arcella cf. *lobostoma*	**P**	0	1	0	1	1	0	0	0
	Arcella a	**P**	2	2	2	2	1	2	0	1
	Arcella b	**P**	1	1	2	1	1	2	0	1
	Arcella c	**P**	0	1	1	0	1	0	0	0
CENTROPYXIDAE										
	Centropyxis	**P**	0	0	0	0	1	0	0	0
TRIPONOPYXIDAE										
	cf. Cyclopyxis	**P**	0	0	0	0	0	1	0	0
DIFFLUGIIDAE										
	Difflugia cf. *elegans*	**P**	0	0	0	1	0	0	0	0
	Difflugia gramen	**P**	0	0	0	0	1	0	0	0
	Difflugia cf. *manicata*	**P**	0	0	0	1	0	0	0	0
EUGLYPHIDAE										
	Euglypha	**P**	0	0	0	1	0	0	0	0
	indet. testate	**P**	0	0	0	0	0	0	1	0
ROTIFERA										
BDELLOIDEA										
	indet. contr. 1 [8 main unci]	**P**	0	0	0	1	0	0	0	0
	indet. contr. 2 [2 main unci]	**P**	1	1	1	2	2	2	1	1
	indet. contr.3 [3 main unci]	**P**	0	2	1	1	1	2	1	1
MONOGONONTA										
ASPLANCHNIDAE										
	Asplanchna cf. *brightwelli*	**P**	0	0	0	0	0	0	0	1
BRANCHIONIDAE										
	Anuraeopsis navicula	**P**	0	0	0	0	0	0	0	1
	Keratella procurva	**P**	2	2	3	3	3	2	0	3
	Brachionus cf. *lyratus*	**P**	0	0	1	0	0	0	0	0
	Brachionus cf. *leydigi*	**P**	1	0	0	0	0	0	0	0
	Brachionus quadridentatus	**P**	0	2	2	1	1	1	1	0
DICRANOPHORIDAE										
	Encentrum cf. *uncinatum*	**P**	0	0	1	0	0	0	0	0
EPIPHANIDAE										
	Microcodides robustus	**P**	1	0	0	0	0	0	0	0
EUCHLANIDAE										
	Euchlanis cf. *dilatata*	**P**	1	2	1	1	1	1	0	1
	Euchlanis cf. *meneta*	**P**	3	0	3	3	3	3	1	1
GASTROPODIDAE										
	Ascomorpha ecaudis	**P**	0	0	0	0	0	1	0	0
	Ascomorpha saltans	**P**	1	0	1	1	1	1	0	0

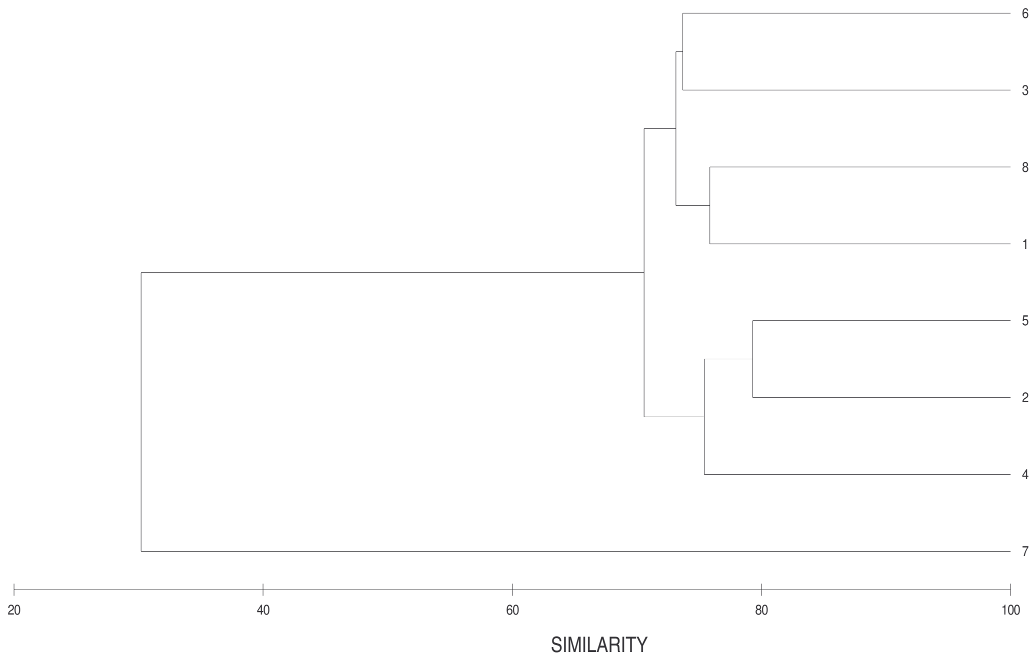
TAXA		SITE								
		P/T	*1*	*2*	*3*	*4*	*5*	*6*	*7*	*8*
LECANIDAE										
	Lecane bulla	**P**	0	0	0	0	0	1	0	0
	Lecane curvicornis	**P**	2	3	2	3	3	3	0	1
	Lecane cf. *eylesi*	**P**	1	0	0	0	0	0	0	0
	Lecane cf. *hamata*	**P**	0	0	0	0	0	1	0	1
	Lecane luna	**P**	1	0	1	0	0	1	0	0
	Lecane sp. A	**P**	0	0	0	0	0	0	1	0
LEPADELLIDAE										
	Colurella uncinata bicuspidata	**P**	0	1	0	2	1	1	0	0
	Lepadella biloba	**P**	1	0	1	2	1	2	0	2
	Lepadella patella	**P**	0	1	0	0	0	2	0	0
MYTILINIDAE										
	Lophocharis salpina	**P**	0	0	0	0	0	0	2	0
NOTOMMATIDAE										
	Cephalodella catellina	**P**	0	0	1	0	1	0	0	0
	Cephalodella euderbyi	**P**	0	0	0	2	0	0	0	0
	Cephalodella gibba	**P**	2	1	2	2	2	2	1	2
	Cephalodella megalocephala	**P**	0	0	1	0	0	0	0	0
	Cephalodella cf. *pachdactyla*	**P**	0	0	0	0	1	0	0	0
	Cephalodella ventripes	**P**	2	2	2	3	3	1	0	2
	Cephalodella sp. nov.	**P**	0	0	1	0	0	0	0	0
	Cephalodella sp. A	**P**	0	0	0	0	0	0	1	0
	Eosphora anthadis	**P**	0	0	0	1	0	0	1	1
	Notommata sp. A	**P**	0	0	0	0	0	0	1	0
SCARIDIIDAE										
	Scaridium longicaudum	**P**	1	1	1	1	2	2	0	1
SYNCHAETIDAE										
	Polyarthra dolichoptera	**P**	1	0	1	0	0	0	0	2
TESTUDINELLIDAE										
	Testudinella patina	**P**	0	0	0	0	0	1	1	0
TRICHOCERCIDAE										
	Trichocerca rattus	**P**	1	0	1	2	2	2	0	0
	Trichocerca vernalis	**P**	2	0	0	0	1	1	0	1
	Trichocerca	**P**	0	0	0	0	0	0	1	0
PLATYHELMINTHES										
TURBELLARIA	Turbellaria spp.	**P**	0	0	1	0	0	0	0	1
ANNELIDA										
OLIGOCHAETA	Oligochaeta spp	**P**	2	0	2	2	2	2	0	2
MOLLUSCA										
PLANORBIDAE	?*Isidorella* sp.	**P**	0	0	0	0	0	0	1	0
CRUSTACEA										
COPEPODA										
CALANOIDA										
	Boeckella triarticulata	**P**	0	0	0	0	0	0	2	1
	Calamoecia baylyi	**P**	0	0	0	0	0	0	3	1
CYCLOPOIDA										
	Mesocyclops sp.	**P**	1	2	1	1	2	1	0	3
	Metacyclops sp. A	**P**	1	2	1	2	2	1	1	1
	Metacyclops sp. B	**P**	0	0	0	0	0	0	0	0
	copepodites	**P**	3	3	3	3	3	3	0	3
	nauplii	**P**	4	3	3	3	3	3	3	4

TAXA			SITE							
		P/T	***1***	***2***	***3***	***4***	***5***	***6***	***7***	***8***
HARPACTICOIDA										
	cf. *Elaphoidella*	**P**	1	0	0	0	0	1	0	0
CLADOCERA										
CHYDORIDAE										
	Alona rigidicaudis	**P**	1	0	0	0	0	1	0	0
	Alona sp. A	**P**	0	0	0	0	0	0	1	0
DAPHNIIDAE										
	Ceriodaphnia sp.	**P**	0	0	1	0	0	0	0	0
MOINIDAE										
	Moina cf. *micrura*	**P**	2	1	2	1	0	1	1	1
OSTRACODA										
	cf. *Australocypris*	**P**	1	1	1	1	2	1	0	1
	cf. *Alboa*	**P**	1	1	1	0	2	1	1	2
	cf. *Cypretta*	**P**	0	1	1	1	1	1	0	1
	cf. *Stenocypris*	**P**	0	0	0	0	0	1	0	0
	indet. juveniles	**P**	0	0	0	0	0	0	3	0
ANOSTRACA										
	Branchinella occidentalis	**P**	0	0	0	0	0	0	2	0
	Branchinella frondosa	**P**	2	2	2	2	0	3	0	2
CONCHOSTRACA										
	Eulimnadia dahli	**P**	2	2	3	3	3	3	0	2
	Caenestheria sp.	**P**	0	0	0	0	0	0	4	0
NOTOSTRACA										
	Triops (a) australiensis	**P**	0	0	0	0	0	0	3	0
INSECTA										
COLLEMBOLLA										
SYMPHYPLEONA	Symphypleona spp.	**T**	1	1	1	1	0	1	1	0
DIPTERA										
	Unknown diptera (larvae)	**T**	0	1	1	0	1	1	1	2
CHIRONOMIDAE	Chironomid spp. (pupae)	**T**	1	2	1	1	2	2	0	0
Chironominae	*Polypedilum* sp. (?M1 Cranston)	**T**	0	0	2	1	2	2	1	0
	Chironomus aff. *alternans*	**T**	2	0	0	2	2	2	0	0
	Chironomus tepperi	**T**	0	0	0	0	2	0	2	0
	Tanytarsus sp.	**T**	3	0	0	0	0	2	0	0
	Parachironomus sp. (?K2 Cranston)	**T**	0	0	0	0	0	1	0	0
	Paraborniella tonnoiri	**P**	1	3	2	2	2	3	0	2
	Chironiminae early instars	**T**	3	0	2	3	3	4	0	4
Tanypodinae	*Procladius* sp.	**T**	0	0	1	1	0	2	0	0
	Paramerina sp.	**T**	0	0	0	0	2	0	0	0
	Ablabesmyia ?*notabilis*	**T**	0	0	0	0	2	3	0	0
	Larsia ?*albiceps*	**T**	2	0	1	0	0	2	0	0
Orthocladiinae	*Orthocladiinae* nr *V44*	**T**	2	2	2	1	1	2	0	2
CULICIDAE	Culicidae spp. (pupae)	**T**	3	3	2	3	1	3	2	2
	Culex (Cx) australicus	**T**	3	4	2	3	2	2	0	3
	Aedes (Psk.) bancroftianus	**T**	2	3	0	2	3	3	0	0
	Aedes (Och.) normanensis	**T**	2	0	0	0	0	0	2	0
	Aedes (Och.) ENMs sp. #71	**T**	0	1	1	0	0	0	1	0
CERATOPOGONIDAE	Ceratopogonid spp. (pupae)	**T**	0	0	0	0	0	0	0	0
	Ceratopogoninae spp.	**T**	2	2	0	1	2	0	0	3
	Dasyheleinae spp.	**T**	0	1	0	1	1	0	0	1
	Forcipomyiinae spp.	**T**	0	0	0	0	0	1	0	0
TIPULIDAE										

TAXA			SITE							
		P/T	1	2	3	4	5	6	7	8
	Tipulidae spp.	T	0	2	2	1	2	2	0	1
DOLICHOPODIDAE										
	Dolichopodidae spp.	T	0	0	1	0	0	0	0	1
THAUMALEIDAE										
	?Thaumaleidae spp.	T	1	0	1	0	0	0	0	0
ANISOPTERA										
	Anisoptera spp. (juveniles)	T	0	0	0	0	2	2	0	0
EPHEMEROPTERA										
BAETIDAE	*Cloeon* sp.	T	0	0	0	2	1	0	0	0
LEPIDOPTERA										
PYRALIDAE	Nymphulinae spp. (larvae)	T	0	0	0	0	1	0	0	2
HEMIPTERA										
CORIXIDAE										
	Agraptocorixa sp. (female)	T	0	0	0	1	0	0	0	0
	Corixidae spp. (juveniles)	T	0	0	0	0	0	1	0	0
NOTONECTIDAE										
	Anisops sp. (female)	T	0	0	0	0	1	0	0	0
COLEOPTERA										
DYTISCIDAE										
	Allodessus bistrigatus	T	0	1	0	0	0	0	0	0
	Antiporus sp. (larvae)	T	1	0	0	0	0	0	0	0
	Copelatus ferrugineus	T	0	2	0	0	0	1	0	0
	Copelatus sp. (larvae)	T	1	2	2	2	2	3	0	0
	Cybister sp.	T	0	0	0	0	1	0	0	0
	Eretes australicus	T	0	0	1	1	0	0	0	0
	Onychohydrus scutellaris	T	0	0	0	0	1	0	0	0
	Paroster sp. (larvae)	T	1	0	0	2	0	0	0	2
	Tribe Bidessini (larvae)	T	3	2	2	2	1	2	0	2
HYDROPHILIDAE										
	Berosus nutans	T	0	0	0	0	0	0	1	0
	Helochares sp. (larva)	T	0	0	0	0	0	0	0	1
SCIRTIDAE										
	Scirtidae spp. (larvae)	T	2	2	2	2	0	3	0	0
STAPHYLINIDAE										
	Staphylinidae spp. (T)	T	0	1	0	0	0	0	0	0
	Total number of 'species'		***50***	***43***	***55***	***55***	***56***	***62***	***33***	***46***

Appendix 4. Classification dendograms.

Cluster analysis of total presence/absence data set for permanent resident fauna



Cluster analysis of total presence/absence data set for temporary resident fauna