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Effects of the Ok Tedi Copper Mine on the Benthic Macro-invertebrate Fauna of Forest-Fringed Oxbow Lakes of the Fly River System, Papua New Guinea

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ABSTRACT

The Fly River, southern Papua New Guinea, is one of the worlds largest river systems. Although the upper catchment extends to over 4000 m altitude, the majority of the drainage basin consists of an extensive floodplain. Oxbow lakes, which are permanently inundated, form an important habitat of the floodplain, acting as drought refuges for many fish species. Aquatic macroinvertebrates comprise a significant component of the diet of some of these species. Given their importance and sensitivity to changes in water quality, it was crucial to determine if a major mining operation in the headwaters was affecting the macroinvertebrate fauna. The present study was designed to detect effects, if any, of the Ok Tedi copper mine on the benthic macroinvertebrate fauna of forest-fringed oxbow lakes downstream of the mine.

Macroinvertebrates were collected by replicate Ekman grab samples from three control and three potentially-impacted oxbow lakes. Physico-chemical parameters (temperature, pH, dissolved oxygen, and concentrations of dissolved and benthic particulate copper) were measured at each site and differences in benthic macroinvertebrate assemblages were related to differences in the quality of both the water and sediment. Concentrations of dissolved and benthic particulate copper at sites downstream of the mine were elevated, however, there were no statistically significant differences in the benthic fauna which could be related to the presence of effluent from the mine. The benthic fauna of all lakes investigated was depauperate, with anoxia at the water-sediment interface a likely determining factor, although naturally high rates of sediment deposition could also be limiting the fauna.

Key Words: Wetlands, Aquatic ecology, Tropical ecology, Mining impacts, Copper, Sediment.

INTRODUCTION

The Fly River, southern Papua New Guinea, is one of the world's great river systems. It

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arises in the karst Star Mountains of New Guinea at approximately 4000 m altitude and is over 1200 km in length. Within 200 km of the headwaters, the river descends to an extensive, seasonally-inundated floodplain across which it meanders to the Gulf of Papua; with a fall of approximately 20 m over 800 km. The floodplain is covered in primary rainforest in the upper reaches, tall grasses and wild sugar-cane in the middle reaches and mangrove swamps towards the estuary. Annual rainfall in the headwaters is in excess of 10 m, and 3-4 m on the floodplain, resulting in a discharge of 6 000 cumecs, making it comparable to the Niger, Danube or Indus rivers (Welcomme 1985, Smith and Bakowa 1994). However, with a catchment of 76 000 km^2, the Fly River has one of the highest discharges per unit area of catchment in the world.

The floodplain in the middle reaches of the Fly River is comprised of four broad habitat types: blocked valley lakes (total area 245 km^2) where the Fly River acts as an "hydraulic dam" to back-flood tributaries to form broad, shallow lagoons; oxbow lakes (122 km^2) formed from cut-off meander bends of the river; and, grassed and forested floodplain (2473 km^2 combined) (Smith and Bakowa 1994). Utilisation of these habitats by fish has been reported by Smith and Bakowa (1994), and in general terms, they support an extremely productive and diverse fish fauna (Smith and Bakowa 1994). Although a minor habitat in surface area, oxbow lakes are important as they are permanently inundated and provide a drought refuge for species of fish restricted to the floodplain; during the El Ninõ drought of 1993, all floodplain habitats except the deep oxbow lakes were dry (AWS pers. observ.). The dominant fish feeding guild on the floodplain are aquatic insectivores (Roberts 1978, Kare 1991, OTML 1997). Any factors that could adversely affect the quality of this habitat therefore could have significant effects on populations of these fish species, either directly, through degraded water quality, or indirectly through their trophic dependencies on aquatic invertebrates. This would be particularly important during drought periods when habitat area is reduced to these deeper, permanent oxbow lakes.

Located on the headwaters of the Ok Tedi, a major tributary of the Fly River ('ok' means river in the local Yonggom language; Smith and Bakowa 1994), is the Ok Tedi Copper Mine, one of the worlds largest copper producers. Construction for the project commenced in 1981. In 1984 the mine commenced operations as a gold producer, before switching to the production of copper concentrate in 1988. Production is expected to continue until 2010. Due to the combination of high rainfall, frequent seismic activity and geological instability, construction of a tailings dam was not considered feasible and, therefore, the mine operated without waste retention (Smith and Bakowa 1994). Currently, in excess of 60 M tonnes of tailings and waste rock (approximately 50:50) enter the river system each year resulting in significant bed aggradation in the upper (5 -10 m) and middle reaches (3 m). Since gold production ceased, copper is regarded as the major metal of environmental concern (Smith and Bakowa 1994).

Given the location, scale and mode of operation of the project, an extensive environmental study was initiated in 1981 (Maunsell and Partners 1982), with permanent hydrological, chemical and biological monitoring programs established in 1983 (Smith and Bakowa 1994). Biological monitoring has concentrated on the fish fauna (*viz* species diversity, biomass and metal uptake) to satisfy the State's criteria of maintaining a viable, subsistence fishery. However, there have also been a number of studies of the aquatic invertebrate fauna of upland streams, lowland rivers and floodplain habitats in the system (see review by Storey and Maie 1993). The most recent was by the Australian Centre for Tropical Freshwater Research (ACTFR) which conducted a preliminary survey of macroinvertebrate assemblages of floodplain habitats (Benson and Pearson 1994). The study evaluated the suitability of different sampling methodologies and assessed habitat utilisation. The project detected significant differences in macroinvertebrate communities within and between different floodplain habitats (Benson and Pearson 1994).

Within habitats, the benthic fauna appeared depauperate, with low abundance and species richness compared to a relatively rich fauna on marginal submerged aquatic macro-phytes (Benson and Pearson 1994). The sparseness of the benthic fauna was attributed to low dissolved oxygen (DO) levels, although other factors, such as seasonality of the shallower habitats was also hypothesised as a possible cause. The report stated, however, that the real reasons for the paucity of the benthic fauna remain unresolved (Benson and Pearson 1994).

Therefore, the possibility existed that mine-derived sediments had entered the floodplain waterbodies, settled to the bottom and had adversely affected the benthic fauna. The present study was therefore implemented with the aim of determining if the depauperate benthic fauna reflected a mine-effect or limitation due to some natural factor, such as low DO.

STUDY AREA AND SITE SELECTION

In order to minimise habitat differences, sampling was restricted to oxbow lakes which were located within primary rainforest and joined to the main river via a tie-channel, allowing interchange of water between the river and the oxbow. A total of six sites were selected; three control and three sites downstream of the mine. Downstream sites were the Ok Tedi Oxbow on the lower Ok Tedi, immediately upstream of where the Ok Tedi joins the Fly River at D'Albertis Junction, Kuambit Oxbow at Kuambit, approximately 2 km downstream from D'Albertis Junction, and Oxbow 2 approximately 40 km downstream from the junction (Figure 1). Two control sites were selected on the Fly River upstream of D'Albertis Junction, Moian and Drimdenasuk Oxbows, 37 and 82 km above the junction respectively. The third control, Oxbow 8, was a forested oxbow on the middle reaches of the Strickland River, the major tributary of the Fly River (Figure 1).

METHODS

A stratified sampling design was used within each oxbow to allow for differences in water depth. Three locations were sampled; one near the edge (shallow), the second between the centre of the lake and edge (intermediate), and the third towards the centre of each lake (deep, although not necessarily the deepest point in the wetland). All locations were located approximately mid-way along the longitudinal axis of each oxbow. Benthic fauna was sampled by Ekman grab (surface area of 600 cm^2). Ten replicate samples were haphazardly taken at each of the three locations, giving 30 replicates per site. Samples were stored in 70% alcohol in plastic bags and returned to the laboratory for processing.

Physico-chemical measurements were taken at all sites. Temperature (°C), dissolved oxygen (DO, percent saturation), and pH were measured using a YSI multi-probe Water

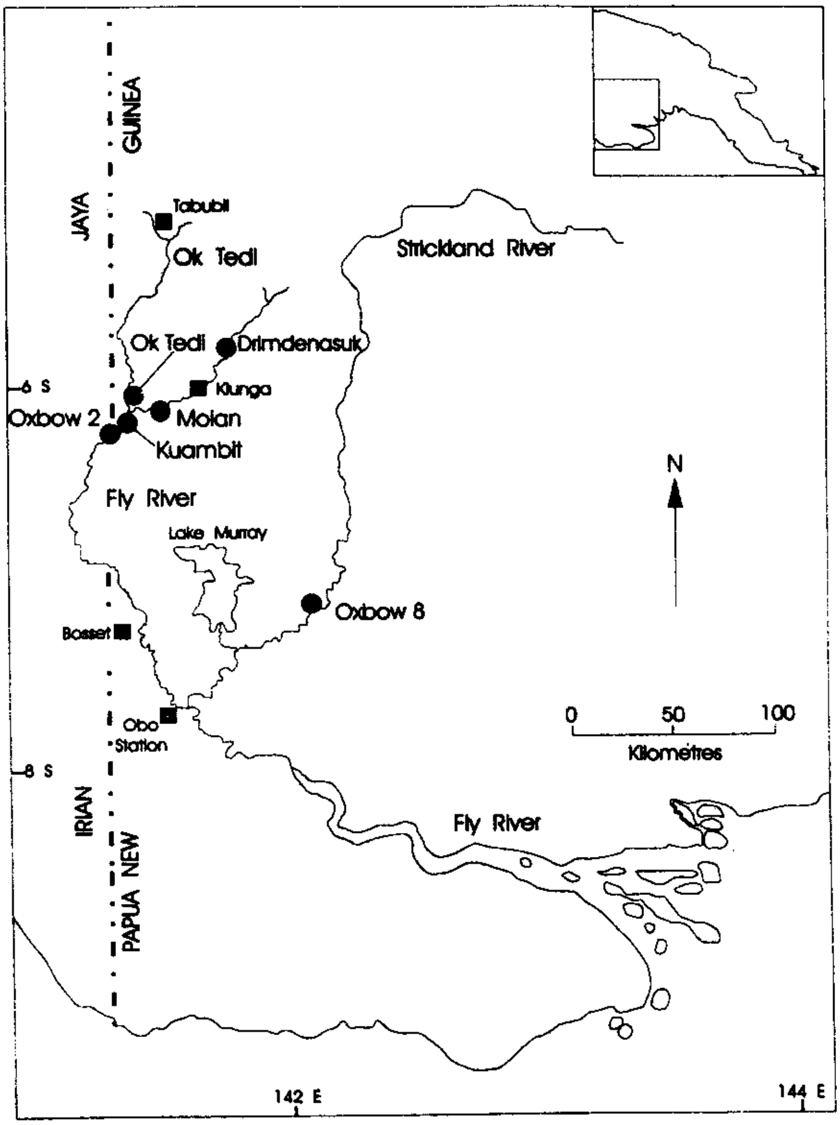

Figure 1. Location of the Fly River in Papua New Guinea, indicating position of the forested oxbow lakes sampled for benthic macroinvertebrates.

Quality Monitor. Vertical profiles of temperature and DO were taken at the deepest location, with measurements recorded at 50 cm intervals from the surface to the bottom. Particulate copper (pCu) concentrations in benthic sediments were determined from a sediment sample taken from the surface layer of the first Ekman grab sample from each location at each site (n=3). Samples were removed from grabs using a plastic spatula and placed in "whirl-packs" for return to the laboratory. Surface and bottom samples of water for analysis of dissolved copper (dCu) concentrations were taken from each location. Surface samples were collected directly into acid washed bottles, bottom samples were taken using a 1.7 litre General Oceanics Niskin water sampling bottle (Model 1010); at each location the sampler was lowered to the lake bottom, raised several centimetres and then triggered. Samples were then transferred to acid washed bottles. All sampling was conducted between the 5th and 11th July 1994; coinciding with the end of the wet season.

In the laboratory, grab samples for benthic invertebrates were washed through 1 mm and 250 μm mesh sieves and all invertebrates removed under low-power binocular microscope. Taxa were then identified to operational taxonomic units (OTUs) and enumerated; Chironomidae and Ceratopogonidae were not identified beyond family level.

DCu and pCu analyses were conducted by the Environmental Chemistry Laboratory of Ok Tedi Mining Limited (OTML). Water samples for dCu were passed through a 0.45 μm filter before being analysed on a Perkin - Elmer 4100. Benthic pCu samples were acid-digested by microwave before being analysed on a Varian AA 300/400 spectrophotometer.

Analysis of variance (ANOVA) was used to identify significant between-site differences in invertebrate community descriptors (*viz* taxa richness and abundance of animals) and physico-chemical parameters, using all replicates taken within each site. Contrast statements were applied in each analysis to compare control to downstream sites. Prior to analysis, numbers of animals, number of taxa and pCu data were log(x+1) transformed to achieve homoscedasticity and normalise distributions in the majority of cells. Tukey's (HSD) multiple range tests were used to distinguish between levels of main factors where significant differences were found. For temperature, dissolved oxygen and dCu, only near-bed measurements were used, since these values represented the ambient conditions for the benthic fauna. Temperature and oxygen profiles were plotted for each oxbow to detect any stratification. All analyses were performed using SAS version 6 (SAS 1985).

Between-site differences in benthic macroinvertebrate community structure (taxa presence/absence and log(x+1) transformed data) were examined using the Pattern Analysis Package (PATN, Belbin 1995). Samples were classified into groups using an agglomerative hierarchical fusion technique (flexible unweighted pair group arithmetic averaging (UPGMA)), and ordinated using semi-strong hybrid multidimensional scaling (SSH). Dissimilarity matrices were derived using the Bray-Curtis association measure. Gradients in physico-chemical variables through ordination space were calculated using principal axis correlation (PCC), with the significance of gradients tested against Monte Carlo randomisations (n=100) of the data (MCAO) (Belbin 1995).

RESULTS

Physico-chemistry

ANOVAs detected significant between-site differences in pCu and dCu (Table 1, Figure 2), with contrast statements showing significantly higher concentrations of pCu and dCu at downstream compared to control sites (Table 1). The three individual downstream sites had significantly higher benthic pCu concentrations than the three control sites. However,

Table 1. Between-site differences in physico-chemical parameters as determined by ANOVA. Sites are arranged in descending order and sites underlined by a common line are not significantly different at $\alpha < 0.05$ (Tukey's (HSD) Multiple Range Tests). Control sites (lower case): Drimdenasuk = drm, Moian = moi, Oxbow 8 = ox8; Downstream sites (upper case): Ok Tedi = OKT, Kuambit = KUA, Oxbow 2 = OX2). Contrast statement: C = control; D = downstream; Significance levels: $p > 0.05$, ns; $p < 0.05$ *; $p < 0.01$, **; $p < 0.001$, ***).

Effect	df	F	P	Tukey's Range Test						Contrast
pCu	5,17	165.80	***	OKT	KUA	OX2	ox8	moi	drm	D>C ***
dCu	5,17	9.62	***	KUA	OX2	OKT	ok8	moi	drm	D>C ***
Temp	5,17	30.19	***	ox8	drm	OKT	OX2	KUA	moi	C>D **
pH	5,17	4.97	*	moi	OKT	OX2	drm	OX8	KUA	C=D n.s.
DO	5,17	18.12	***	moi	drm	ox8	OKT	OX2	KUA	C>D ***

dCu did not show the same pattern; Kuambit (mean 22.2 $\mu g\ L^{-1}$) and Oxbow 2 (mean 20.3 $\mu g\ L^{-1}$) were significantly higher than the control sites (Drimdenasuk, 1.3 $\mu g\ L^{-1}$; Moian, 1.4 $\mu g\ L^{-1}$; Oxbow 8, 2.2 $\mu g\ L^{-1}$); the Ok Tedi oxbow had an intermediate concentration (8.8 $\mu g\ L^{-1}$) (Table 1, Figure 2).

Within downstream sites, concentrations of dCu were generally higher in near-bottom as compared to surface water samples (surface concentrations of dCu in Ok Tedi, Kuambit and Oxbow 2 were 9.7, 4.5 and 14.0 $\mu g\ L^{-1}$ respectively).

Significant between-site differences in temperature, pH and DO were detected (Table 1). Benthic waters in Oxbow 8 had significantly higher temperatures compared to all other sites (Figure 2); this site was shallower than other sites (4.0 m, compared to mean of 6.2 m) which may have accounted for this difference. Tukey's (HSD) range test did not detect any significant differences in temperature amongst the other sites, however, the contrast detected significantly higher temperatures in control compared to downstream sites (Table 1). However, this result was overly influenced by the higher temperatures in Oxbow 8.

Moian had a significantly higher pH than both Oxbow 8 and Kuambit, but with no other between-site differences. Contrasts did not show a difference in pH between control and downstream sites (Figure 2, Table 1). Highest benthic DO concentrations were recorded in the three control oxbows; Moian, Drimdenasuk and Oxbow 8. DO concentrations in Moian and Drimdenasuk were higher than all downstream sites, with levels in Oxbow 8 higher than in Oxbow 2 and Kuambit (Table 1). Contrasts showed a significant difference, with higher benthic DO levels in control compared to downstream sites.

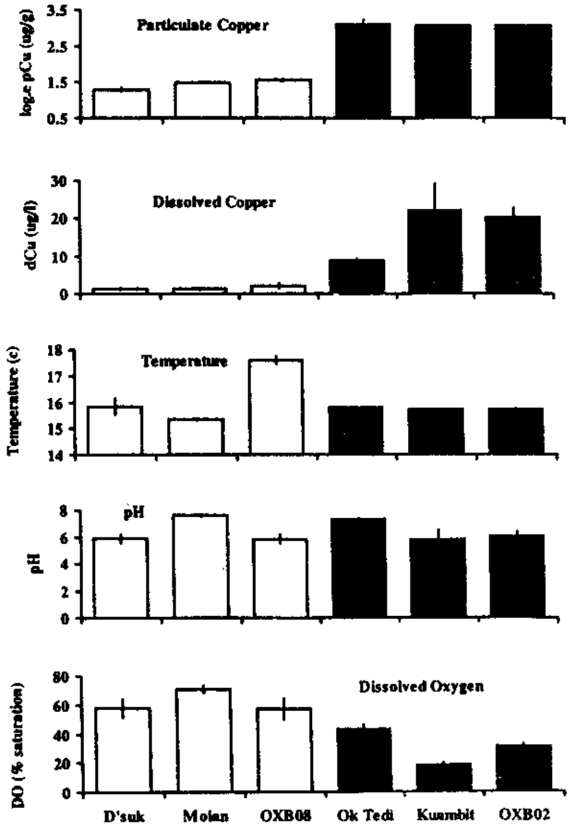

Figure 2. Between site differences in physico-chemical parameters (mean ± 1 SE)(open bars = control sites, shaded bars = downstream sites).

Temperature profiles for the deepest location sampled in each wetland showed little vertical change. The greatest range from top to bottom was 2.5°C in the Ok Tedi oxbow. Other sites showed approximately a 2°C change in temperature from top to bottom, with most of the change occurring in the top 1-2 metres. Vertical stratification of temperature with a thermocline was not evident. DO concentrations generally decreased from surface maxima to bottom minima. Moian was an exception, with top and bottom DO about 70%. Drimdenasuk changed from 82% at the surface to 69% at the bottom with a mid-water minimum of 50%. DO in Oxbow 8 decreased from 84% at the surface to 43% at the bottom, Ok Tedi decreased from 54 to 41%, Kuambit from 43% at the surface to 21% at the bottom, and Oxbow 2 from 59 to 32%.

Macroinvertebrate Assemblages

A total of 1430 individuals belonging to 27 taxa of macroinvertebrates were collected from the 180 grab samples. Oxbow 8 contained 63% of the total number of animals found; 904 individuals from 13 taxa, comprised mainly of Nematoda (370 individuals) and Gastropoda sp.H (369 individuals). Kuambit had the next highest abundance with 130 individuals from six taxa, followed by Moian with 115 individuals from five taxa, Oxbow 2 with 110 individuals from four taxa, Drimdenasuk with 106 individuals from 13 taxa, and Ok Tedi with 65 individuals from six taxa. The Nematoda, followed by Chironomidae and Ceratopogonidae were the most commonly-occurring taxa, found at all sites, and the Nematoda, followed by Gastropoda, Chironomidae and Ceratopogonidae were most numerically abundant, comprising 90% of the fauna.

Table 2. Between-site differences in (a) number of taxa and (b) number of animals as determined by ANOVA. Sites are arranged in descending order and sites underlined by a common line are not significantly different at $\alpha < 0.05$ (Tukey's (HSD) Multiple Range Tests) (Control sites (lower case); Drimdenasuk = drm, Moian = moi, Oxbow 8 = ox8, Downstream sites (upper case); Ok Tedi = OKT, Kuambit = KUA, Oxbow 2 = OX2) (contrast statement, C = control; D = downstream) (significance levels; p > 0.05, ns; p < 0.05, *; p < 0.01, **; p < 0.001, ***).

Effect	df	F	P	Tukey's Range Test						Contrast
(a)	5,179	17.55	***	ox8	KUA	drm	OX2	moi	OKT	C>D ***
(b)	5,179	26.66	***	ox8	KUA	OX2	moi	drm	OKT	C>D ***

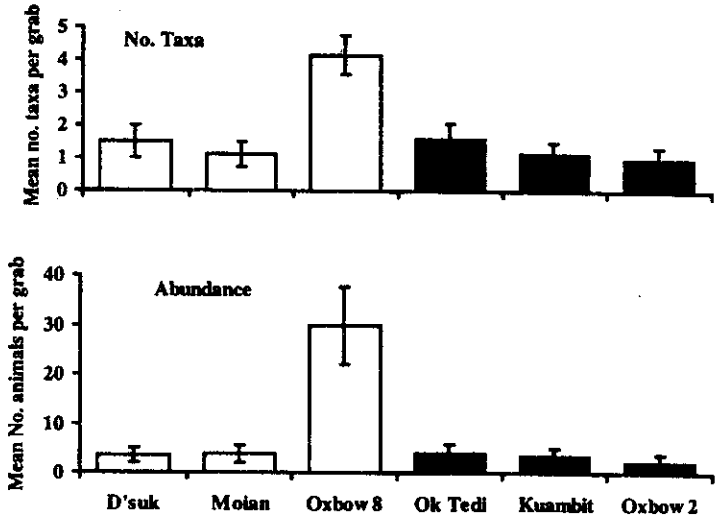

Figure 3. Between site differences in number of taxa and abundance of benthic macroinvertebrates in grab samples (mean ± 1 SE) (open bars = control sites, shaded bars = downstream sites).

ANOVAs showed significant site differences in the number of taxa and abundance of animals, with Oxbow 8 having significantly more taxa and greater abundance of animals than any other site, but with no significant differences amongst the remaining sites (Table 2, Figure 3). Contrasts showed significantly higher number of taxa and abundance of animals in control compared to downstream sites. These analyses were strongly influenced by Oxbow 8. Repeating the analyses with this site removed failed to detect any between-site differences by ANOVA and, similarly, contrasts were no longer significant.

UPGMA classification and SSH ordination were performed on the mean abundances of taxa for all replicate grabs taken from each site (n=6). Condensing data to characterise the fauna within each site was necessary due to the sparse nature of the fauna within individual grab samples. Analysis on presence/absence and log(x+1) transformed data failed to separate control from downstream sites on benthic macroinvertebrate assemblages (Figures 4 and 5). Classifications separated Oxbow 8 as an "outlier" at the first division, with subsequent divisions showing no separation of control from downstream (Figure 4). Similarly, Oxbow 8 tended to be an outlier in both ordinations, with no clear separation of remaining control sites from downstream sites (Figure 5).

PCC showed there were no significant gradients in physico-chemical data through either ordination. However, there were significant gradients in number of taxa ($p<0.04$) and abundance of animals ($p<0.01$) through the presence/absence ordination, with the latter in the direction of Oxbow 8 (Figure 5). Abundance of animals ($p<0.02$) was the only significant gradient in the ordination of log. transformed data, and again, this gradient was in the direction of Oxbow 8 (Figure 5).

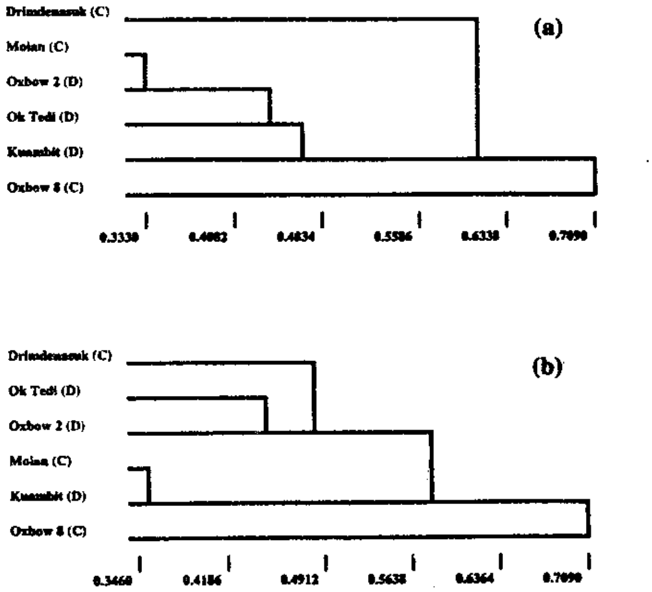

Figure 4. UPGMA classification of oxbow lakes on benthic macroinvertebrate community structure for (a) presence/absence, and (b) log(x+1) transformed data (C = control sites, D = downstream sites).

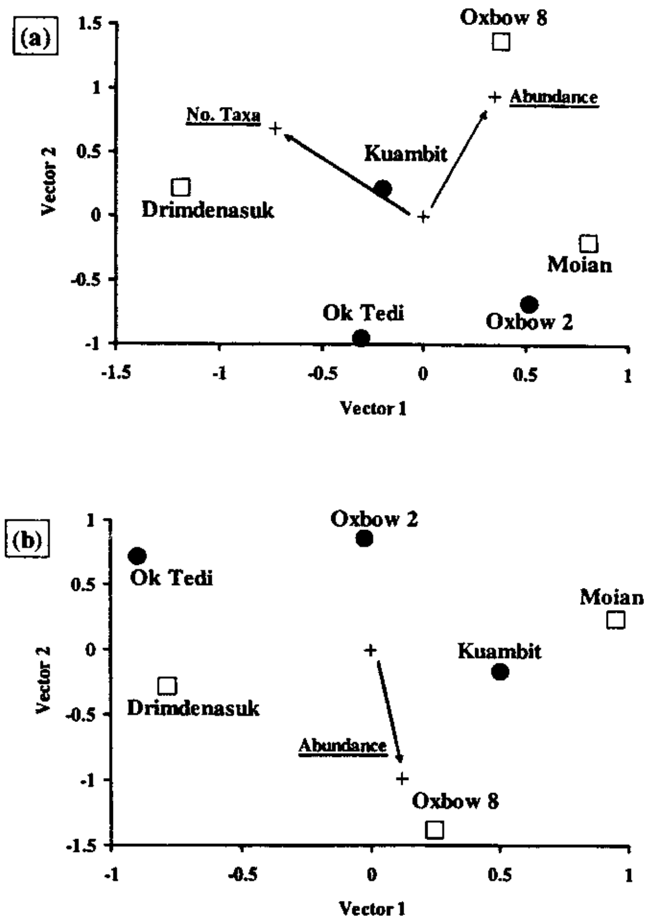

Figure 5. SSH ordinations of oxbow lakes on benthic macroinvertebrate community structure, for (a) presence/absence (an optimum solution was achieved with two dimensions and a stres of 0.16), and (b) log(x+1) transformed data (an optimum solution was achieved with two dimensions and a stress of 0.18). Significant gradients ($p < 0.05$) in community descriptors are indicated (open symbols = control sites, shaded symbols = downstream sites).

DISCUSSION

Oxbow lakes were selected for this study with the intention of maximising the "signal" of the effects of the mine on water and sediment quality in this habitat type so that any effects on benthic macroinvertebrate assemblages would be expected to be at their greatest. A significant mine-effect on water/sediment quality was achieved, with pCu levels two orders of magnitude higher than baseline, and dCu also elevated at downstream sites. Copper levels at downstream sites were representative of concentrations expected in oxbow lakes in this region of the river system receiving mine-derived sediment. In comparison, approximately background levels of dCu (< 3 μg l^{-1}) and pCu (< 50 μg g^{-1}) were recorded from control sites.

Among-site differences in benthic pCu were marked, with elevated levels at the downstream oxbows. Differences in dCu were less clear, with values from Ok Tedi overlapping between control and downstream. This partly reflected small sample size (n=3) combined with large variability in these data, however, concentrations amongst the downstream sites may have been also influenced by pH. It is generally recognised that as pH decreases, there is increased mobilisation and toxicity of metals such as copper (Holdway 1991). Kuambit oxbow and Oxbow 2 had mean benthic pH of 5.8 and 6.1, compared to 7.4 in Ok Tedi oxbow. Mobilisation in Ok Tedi oxbow may have been reduced relative to the others, accounting for the lower dCu concentrations.

In the downstream sites, benthic dCu concentrations were generally higher than surface dCu concentrations. Previous field studies have demonstrated a release of dCu from benthic sediments into pore-water at the immediate (e.g. 1-2 cm) sediment-water interface, resulting in elevated concentrations (OTML/CSIRO 1993).

Reasons for the higher benthic DO concentrations in control compared to downstream sites were not clear. Algal blooms were not evident at any site, and time of day when sampling occurred should not have had an effect; all sites were sampled between 1000 hrs and 1200 hrs, with the exception of Ok Tedi, which was sampled at 1500 hrs. Low DO levels in downstream sites could indicate reduced algal activity either due to direct toxic effects from dCu or possibly reduced photosynthetic activity as a result of higher total suspended sediment concentrations. Conversely, reduced DO concentrations could indicate depletion due to higher microbial activity in downstream sites.

Toxicity testing of Fly River water on algae indicated that concentrations above approximately 20 ppb dCu may reduce algal growth, however, further studies have shown that dissolved organic carbon (DOC) plays a major role in complexing dissolved copper, making it non-bioavailable (Stauber 1995). Water from Bosset Lagoon has a complexing capacity of 40 ppb dCu, whilst water from the river adjacent to Kuambit oxbow, with lower DOC concentrations, has a complexing capacity of 23 ppb dCu (OTML/CSIRO 1993).

The benthic macroinvertebrate fauna in Oxbow 8 was distinctive; this location contained several taxa of Gastropoda, in high abundance, which were not recorded from other oxbows. Although Oxbow 8 is located within the Fly River catchment, it is on the Strickland River which has quite different fluvial characteristics from the Fly River (e.g. shallower, faster flowing and has a higher natural sediment load; Smith et al. 1990). Also it is geographically removed from the other sites, introducing a possible biogeographical factor. This oxbow was selected as a control due to the absence of additional suitable control oxbow lakes on the Fly River upstream of the junction with the Ok Tedi. However, based on these results, this site is probably not a suitable control.

Re-analysis of benthic macroinvertebrate data after excluding Oxbow 8, failed to separate control sites from downstream, indicating there was no detectable impact of the mine on the structure of benthic macroinvertebrate assemblages of forested oxbow lakes. Previous studies of the effects of mines (Armitage and Blackburn 1985, Mackey 1988), and specifically the Ok Tedi mine (OTML 1985a and b) on benthic macroinvertebrate communities have detected impacts. Armitage and Blackburn (1985) reported significant changes in the structure of chironomid communities in a temperate stream receiving elevated zinc concentrations (> 2.00 mg L^{-1}) and Mackey (1988) recorded significant changes in the macroinvertebrate fauna of a subtropical river with elevated Cu (8.3 - 36.0 mg L^{-1}) and Zn (6.3 - 190.0 mg L^{-1}) concentrations. In the Fly River system, OTML (1985a) reported that the invertebrate fauna was depleted for at least 10 km downstream from the mine due to elevated suspended solids discharged during the construction phase (1981-83), and OTML (1985b) noted that the invertebrate fauna in a sub-catchment, the Ok Mani, was rapidly depleted following the initial release of tailings which changed the physico-chemical characteristics of the system.

The present study supported the findings of Benson and Pearson (1994) who noted that the benthic fauna in various floodplain habitats of the Fly River system was depauperate, both in terms of number of taxa and abundance of animals. Similarly the benthos of blocked valley lakes sampled in the mid-1980's was observed to be depauperate (Hortle, unpublished data).

Average density of benthic macroinvertebrates was approximately 50 individuals m^{-2}, except Oxbow 8, which contained approximately 500 individuals m^{-2}. These densities are low compared to elsewhere (Welcomme 1985, Lindegaard 1992). Bonetto et al. (1978, cited in Welcomme 1985) recorded maximal densities during low and high water of 114,000 individuals m^{-2} and 46,500 individuals m^{-2} respectively in the Laguna Totoras on the Paraná River, Argentina. A second lagoon, Laguna La Brava had macrobenthos densities between 17,500 and 95,300 individuals m^{-2} at low water and 1,700 and 57,000 individuals m^{-2} at high water, with highest densities in the sub-littoral as opposed to the centre (Bonetto et al. 1978). Faunal densities of about 2,000 individuals m^{-2} were recorded for soft bottom sediments in Lake Thingvallavatn, Iceland (Lindegaard 1992), densities of 5,000 - 10,000 individuals m^{-2} were recorded from Lake Mikolajskie, Poland (Prejs 1976, cited in Lindegaard 1992), and 54,000 individuals m^{-2} in Marion Lake, Canada (Kajak and Kajak 1975, cited in Lindegaard 1992).

There are several possible reasons for the naturally depauperate nature of the benthic fauna in forest-fringed oxbow lakes, including extremes of DO saturation, temperature and sedimentation. Benson and Pearson (1994) recording benthic DO concentrations of ~ 10 % saturation and hypothesised that DO was the limiting factor. This compares to minimum and maximum benthic values of ~20 and ~ 70% recorded in this study. It is highly likely that values at the lower end of this range would be deleterious to the fauna.

DO depletion may occur on both a diurnal and a seasonal basis in floodplain habitats. Due to logistical constraints this was not investigated in the current study. However, Chambers (1988) recorded early morning benthic DO minima of approximately 15, 40 and 10% from Bosset Lagoon, Lake Daviumbu, and Lake Pangua on the Middle Fly River floodplain in July 1984 (late wet season). It was suggested that low DO levels were due to high benthic microbial activity outstripping re-aeration by photosynthetic production.

There is little evidence for temperature being restrictive in these systems. The current study recorded little vertical change in temperature. Similarly, Chambers (1988) recorded at most, a decrease of 1 - 2°C over the vertical profiles in Bosset Lagoon, Lake Daviumbu, and Lake Pangua. Chambers (1988) suggested that thermal stratification may occur under calm conditions, particularly during the wet season, with stratification less likely in the dry season when shallower conditions prevail. In the sites sampled in the current study, located in tall, dense primary rainforest (compared to open, grassed floodplain), stratification would be more likely to occur in the dry season when factors that would disrupt stratification, such as fluctuations in river level, and inputs from rainfall and surface runoff would be minimal.

Sedimentation is another possible explanation for the depauperate benthic fauna. Direct impacts of sedimentation have been observed in the Fly River system, but this was restricted to fast-flowing upland rivers subjected to massive and continuous artificial sediment loads. The slow-flowing reaches of the Fly River naturally have a very high sediment load (Smith et al. 1990), therefore, historically, floodplain waterbodies connected to the river would receive riverine sediment on a rising hydrograph and it must be expected that the fauna would be adapted to a degree of sedimentation. However, it is possible that these continuous inputs result in deep, unconsolidated benthic sediments to which few taxa adapt.

There are also indirect effects of sedimentation on the fauna, whereby light penetration may be reduced, limiting algal activity, and thereby resulting in low production of autochthonous organic matter. Combined with continual inputs of fresh, nutrient-poor sediment, the benthic 'grazer' element of the fauna may be restricted.

Even though the benthic macroinvertebrate fauna was depauperate, the non-benthic fauna is relatively productive. Benson and Pearson (1994) reported a diverse and abundant macroinvertebrate fauna on submerged marginal aquatic macrophytes, but with few zooplankton in mid-water tows, and, Chambers (1988) recorded a diverse (51 species), although sparse zooplankton fauna. Studies suggest that these components support a diverse and abundant fish fauna. Smith and Bakowa (1994) recorded 56 species of fish from 26 families utilising the floodplain. The most abundant fish species, the herrings (*Nematalosa flyensis* and *N. papuensis*) (Smith and Bakowa 1994) were planktivorous, suggesting that there is a very productive planktonic population. Similarly, aquatic insectivores are the dominant fish feeding guild in each lake type (OTML 1997).

In conclusion, the results from this study did not show changes in the structure of benthic macroinvertebrate assemblages of forested oxbow lakes that could be related to mine-effects. However, a better understanding of the limnological processes acting in these lakes and a better knowledge of the benthic macroinvertebrate fauna is required. From available data it must be concluded that DO is most likely the limiting parameter, although other factors, such as sedimentation also may be contributing. Studies to examine the effects of the mine on non-benthic (i.e. littoral) invertebrate assemblages in these oxbow lakes may be desirable.

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