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Temporal and spatial variations in fish catches in the Fly River system in Papua New Guinea and the possible effects of the Ok Tedi copper mine

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Synopsis

Biological monitoring of fishes in the Fly River system in Papua New Guinea has been carried out in relation to the input into the system of mine wastes from the Ok Tedi copper mine. A total of 86 fish species representing 32 families has been recorded from sites in the main river channel since the commencement of monitoring operations in 1983. Catfish in the families Ariidae (16 spp.) and Plotosidae (9 spp.) were the dominant groups overall, although *Nematolosa* herrings were the most numerous species, forming over 37% of the catch. However, barramundi, *Lates calcarifer*, comprised the greatest biomass, forming over 30% of the overall catch. Fish catch biomass at most sites showed considerable temporal and spatial variation over the period of sampling. However, significant reductions in biomass, ranging from 65% to 96%, were recorded at most sites in the Ok Tedi, middle and upper Fly. The greatest declines in biomass were recorded in the Ok Tedi at sites closest to the mine, although reductions up to 73% were also recorded at sites in the middle Fly. Barramundi, which formed a high proportion of catch biomass at many sites, particularly in the middle Fly, declined in number at most sites following peak numbers in the early 1990's. The main causative factors involved in the overall declines in fish catches, including both mine-related and non-mine-related factors, are discussed. It is concluded that loss of fish habitat through increased river bed aggradation, due to the input of mined waste rock and tailings, is likely to be the main causal factor. However, other mine-related factors, such as elevated levels of dissolved and particulate copper, and other non-mine-related factors, such as introduced species, may also be involved in declining fish catches.

Introduction

The discharge of waste materials from mining operations into river systems is common in temperate and tropical areas of the world, often with adverse effects on fish stocks and aquatic ecosystems through reductions in water quality and changes in physical habitat conditions (Sengupta 1993). Monitoring fishes in rivers provides a valuable tool for assessing the impacts of anthropogenic activities on the aquatic environment and can be used as an indicator of river ecosystem 'health' and in determining ecological response to remedial measures (Fausch et al. 1990, Harris 1995). Consequently, biological monitoring programmes are often a mandatory requirement for determining the impacts of resource development projects on aquatic ecosystems.

In the Western Province of Papua New Guinea, Ok Tedi Mining Limited carries out extensive environmental monitoring of the Fly River system in relation to the operations of the Ok Tedi copper mine, which discharges waste materials directly into the Ok Tedi, a major headwater tributary of the Fly River. The Fly River is one of the largest rivers in Australasia (mean annual discharge $\sim$6000 $m^3 s^{-1}$), with a catchment area of 76 000 km^2, and flows for over 1200 km from its source in the central highlands to the Gulf of Papua. Much of the catchment of the Fly River, particularly in the upper reaches, consists of dense primary tropical rainforest, while in the middle and lower reaches open savannah forest, swamp forest and seasonally-inundated grasslands predominate. The area is sparsely populated, with an average human density of one to two persons per square kilometre.

The Ok Tedi mine is situated on Mount Fubilan in the headwaters of the Ok Tedi River (Figure 1). The mine is one of the largest copper producing mines in the world. Construction for the project began in 1981 and the mine commenced operations in 1984. There have been three operating stages. Stage one (the treatment of gold ores by cyanide extraction) commenced in 1984. Stage two commenced in mid 1987 and consisted of the gold extraction circuit running in parallel with a flotation circuit to produce copper concentrate. Stage three commenced in mid 1988 when the gold circuit was decommissioned and the mine became a copper concentrate producer. Further infrastructure was completed in 1989, enabling the mine to treat 80 000 tonnes of ore per day at peak production.

The area of the mine is subject to very high rainfall (over 10 m annually) and is geologically unstable, situated on a karst landform which is subjected to frequent seismic activity. Attempts to construct a tailings dam in 1984 were abandoned when a landslide buried the foundations. As a result, the mine has been unable to store waste material and has been operating without waste retention, discharging waste rock and tailings into the headwaters of the Ok Tedi, which then carries the waste material downstream into the Fly River system. The main environmental impacts resulting from mining operations arise from the introduction of large amounts of sediment and copper into the river system. Currently, 80 000 metric tonnes per day (tpd) of tailings and 120 000 tpd of waste rock are discharged by the mine.

Environmental monitoring of the Fly River began in 1981, with an expeditionary survey of fish communities in different habitats, metal levels in biota and water quality. Permanent hydrological, chemical and biological monitoring programmes were established in 1983 and monitoring now extends along the length of the river, from the headwaters down to the river delta and into the Gulf of Papua. Monitoring also includes the extensive floodplain associated with the Fly River system, particularly in its middle reaches.

Although the upper catchment extends to altitudes of over 3500 m, the majority of the drainage basin is low-lying and flat, to the extent that the port of Kiunga, which is 800 river km from the coast, is only 20 m above sea-level. The combination of the flat land and high rainfall has resulted in a broad floodplain with extensive shallow lake systems, occupying an area of 4.5 million ha, making it the largest wetland system in the country. The wetlands of the Fly River are highly productive and play a vital role in the ecology of the river system. The wetland fauna includes a diverse and productive fish community, which provides an important food source for human village communities along the river.

Fishes in floodplain habitats and the main river channel are important to the ecology of the system and have been regularly monitored to provide an indicator of the 'health' and productivity of the river system in relation to the possible effects of mine waste discharges. The freshwater fish fauna of the Fly River is the most diverse in the Australasian region, with 128 recorded species (Roberts 1978, Coates 1993). This compares to a total of 70 species recorded from the Sepik-Ramu River, on the north coast of New Guinea. Seventeen species are known only from the Fly basin and thirty or more are known only from the Fly and one or more of the large rivers in central-southern New Guinea. The fishes of the Fly River basin are characterised by the large size of some species and the abundance of endemic fishes that are poorly represented in other parts of the world, particularly the ariid and plotosid catfishes. In most other ways the composition of the freshwater fish fauna is largely determined by its position in the Australasian zoogeographical zone (Roberts 1978, Coates 1993). In common with the northern Australian region, the fish fauna of the Fly River lacks primary freshwater groups of the Ostariophysi (Bishop & Forbes 1991).

Our objective was to monitor fishes at a range of sites in the Ok Tedi River, Fly River and Strickland River from the commencement of mining operations in 1983 to the present time. The results of monitoring fish communities in floodplain habitats of the Fly River system are reported elsewhere (Swales et al. 1999). The relationships between changes in fish stocks in the main river channel and mine waste discharges have also been reported previously (Smith et al. 1990, Smith & Hortle 1991, Smith & Morris 1992). This study is one of the first to examine the long-term changes in fish stocks in

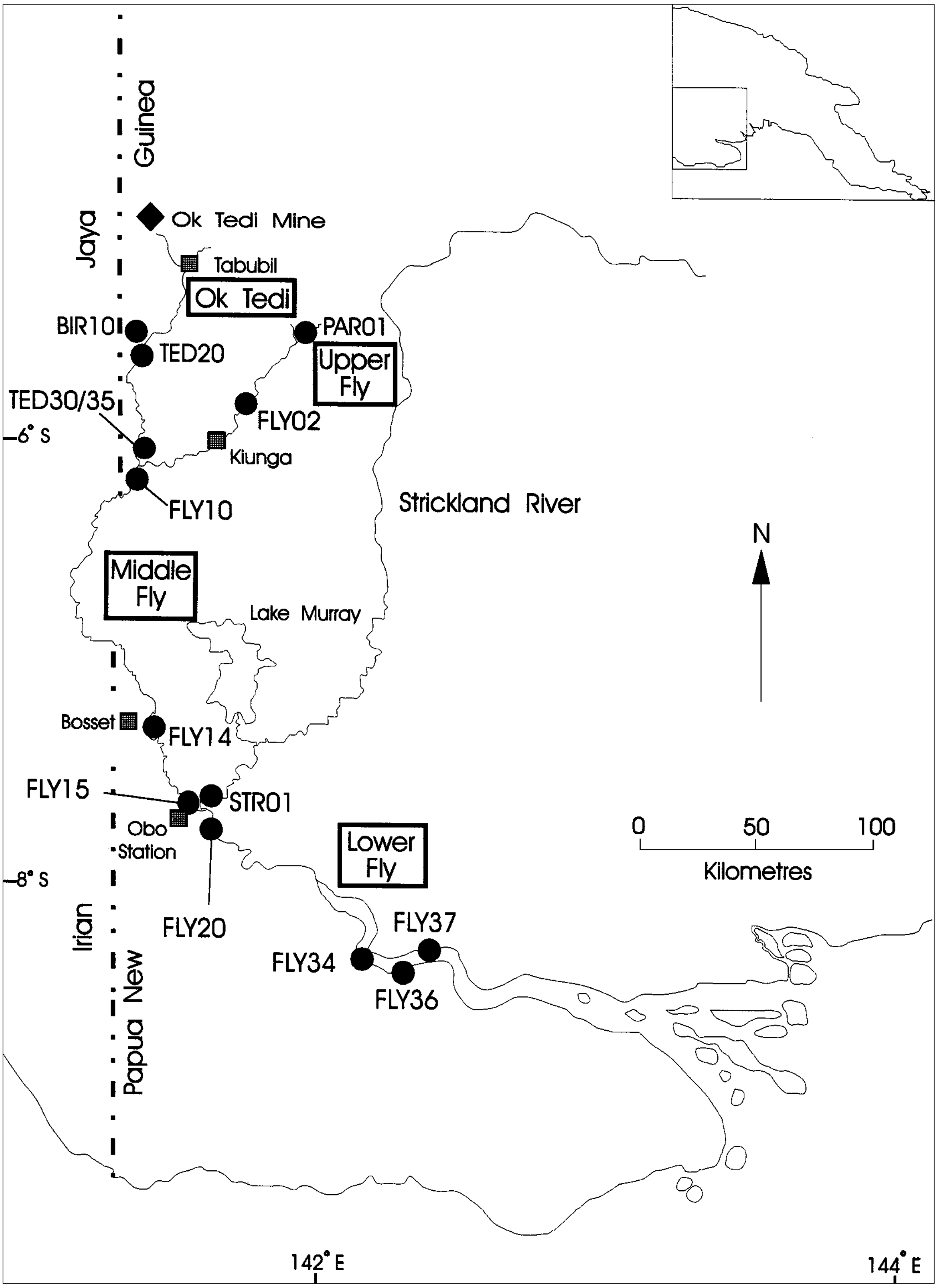

Figure 1. Location of sampling sites in the Fly River system (insert shows location of Fly River catchment in Papua New Guinea).

a major tropical river system in relation to mine waste discharges.

Methods and study sites

Study site description

Sampling was carried out at 13 sites in the Fly River system (Figure 1), including two 'impact' sites in the Ok Tedi, at Ningerum (TED20) and Atkamba (TED30/35), and also at one 'control' site (BIR10), in the Ok Birim, a tributary of the Ok Tedi. Two 'control' sites were sampled in the Upper Fly (PAR01 and FLY02), an area which is unaffected by mine waste discharges, along with three impact sites in the Middle Fly at Kuambit (FLY10), Bosset (FLY14) and Obo (FLY15). A further four potential impact sites were sampled in the lower Fly at Ogwa (FLY20), D'Albertis Island (FLY34), Gilim Island (FLY36) and Sturt Island (FLY37). In addition, a further 'control' site, unaffected by Ok Tedi mine wastes, was situated in the lower Strickland River (STR01).

The Ok Tedi at Ningerum (TED20), 70 km downstream from the mine, is a fast-flowing gravel bed river section of braided channels and rapids, currently overlain by up to 4.5 m of mine-derived sediments. The riparian vegetation is dense primary rainforest. The river channel at the sampling site is approximately 120 m wide with a mean annual flow of 270 $m^3\,s^{-1}$. At Atkamba (TED30), 152 km downstream from the mine, the Ok Tedi is a meandering sand bed river approximately 125 m wide with a mean flow of 920 $m^3\,s^{-1}$. The riparian vegetation is dense primary rainforest. The river bed is currently aggraded by up to 4.5 m with mine derived sediments. In late 1992, due to channel changes associated with a meander cut-off, it was necessary to relocate the original site at Atkamba to a new site (TED35), several kilometres further downstream and closer to the Fly River, and sampling at the previous site (TED30) was discontinued.

The Upper Fly River is similar in size to the lower Ok Tedi and at Kiunga (Figure 1) the river has a mean annual discharge of 1178 $m^3\,s^{-1}$. At Kuambit, situated immediately below D'Albertis junction, where the Ok Tedi joins the Fly, the mean discharge increases to 2161 $m^3\,s^{-1}$. At Obo the mean flow increases to 2244 $m^3\,s^{-1}$, while at Ogwa, below Everill junction, where the Strickland River joins the Fly, the mean flow is 5354 $m^3\,s^{-1}$. The river channel in the middle Fly area is around 2–300 m in width and has a mean water depth of 5–10 m, extending to 30 m in scoured 'pools'. The river bed is currently aggraded by up to 3.5 m with mine-derived sediments. The river channel in the lower Fly area is up to several hundred meters in width and shows considerable tidal influence, with occasional large islands interspersed along the main river channel. Mean monthly water quality parameters recorded at study sites in the Ok Tedi and Fly Rivers during 1996–1997 are summarised in Table 1.

Fish sampling methods

Fishes were sampled by gill-nets and seine-netting, using standardised procedures. Sampling was carried out quarterly at most sites, except TED20 (bimonthly) and FLY02, FLY10 and TED 30/35 (monthly). A standard set of 13 gill-nets, ranging in stretched mesh size from 25 to 175 mm, was used at most sites. The nets were tied at approximately 30° to the bank in a series, each net separated from the next by a large enough distance to allow fish movements between the nets (usually >50 m). Attempts were made to reset nets in the same locations at each site, but this was not always possible because of variability in river levels. Generally, the nets were set in backwater areas, tied to the bank near the interface between the main current and the backwater. Heavy weights were used to anchor the outer end of the nets in position. The order in which nets were set was from larger nets at the downstream end of the series to smaller nets at the upstream end. The nets were set for 24 h and checked at dawn, dusk and the end of the sampling period.

A 50 m seine-net (stretched mesh size 19 mm) was also used at some sites in the Ok Tedi and Fly River to sample shallow bars. Seine-netting was conducted at only five of the thirteen riverine sites (TED20, TED35, FLY02, FLY10 and FLY36). Six replicate hauls were made at each site.

Sample processing

All fish captured were identified to species, measured in length to the nearest 1 mm (standard or total length, depending on body shape) and weighed to the nearest 1 g up to 6 kg and then to the nearest 100 g over 6 kg, and then given away to nearby villagers. When large numbers of a single species were caught in a net, a subsample of 100 specimens was selected at random and measured individually. The remainder were counted and their total weight determined.

Table 1. Summary of mean (standard deviation in parenthesis) monthly water quality parameters at sampling sites in the Ok Tedi and Fly rivers, 1996–1997.

	Sampling site							
	FLY02	TED20	TED35	FLY10	FLY14	FLY15	FLY20	FLY34
Temp °C	26.5 (1.4)	24.8 (0.9)	26.3 (1.1)	26.4 (1.0)	27.8 (1.5)	28.1 (1.4)	27.4 (1.4)	27.7 (3.3)
pH	7.75 (0.46)	8.21 (0.28)	8.01 (0.25)	7.92 (0.28)	7.58 (0.44)	7.65 (0.39)	7.75 (0.34)	7.61 (0.13)
Conductvity $\mu s\ cm^{-1}$	123 (38)	205 (51)	139 (33)	134 (39)	132 (27)	143 (24)	151 (29)	178 (7)
Alkalinity $mg\ l^{-1}$	78 (22)	141 (53)	79 (22)	80 (22)	80 (23)	80 (23)	84 (17)	72 (0)
DO $mg\ l^{-1}$	6.3 (2.2)	8.8 (4.1)	7.2 (3.7)	6.5 (2.0)	5.5 (1.5)	6.0 (2.0)	5.9 (0.9)	7.2 (0.6)
DOC $mg\ l^{-1}$	1.9 (0.7)	2.5 (4.3)	1.3 (0.9)	2.1 (1.1)	2.3 (1.0)	2.5 (0.6)	2.6 (1.2)	3.3 (0.8)
TSS $mg\ kg^{-1}$	42 (44)	2118 (1169)	430 (238)	145 (97)	120 (145)	52 (37)	132 (129)	298 (272)
dCu $\mu g\ l^{-1}$	2.4 (1.5)	16.4 (4.8)	22.8 (9.4)	20.6 (8.1)	19.8 (5.2)	22.0 (6.4)	9.2 (2.9)	10.2 (7.1)
pCu $mg\ kg^{-1}$	194 (248)	1746 (723)	1713 (329)	1396 (227)	1324 (654)	1087 (369)	355 (190)	232 (68)
dZn $\mu g\ l^{-1}$	1.8 (1.3)	2.0 (3.5)	1.4 (0.7)	2.5 (2.2)	2.0 (1.4)	1.5 (0.8)	1.2 (2.6)	0.7 (0.4)
pZn $mg\ kg^{-1}$	123 (75)	204 (104)	223 (85)	253 (119)	229 (74)	208 (97)	232 (106)	154 (7)
dMn $\mu g\ l^{-1}$	3.7 (3.1)	30.6 (11.7)	30.3 (11.0)	18.4 (12.9)	14.2 (10.2)	12.9 (8.2)	2.5 (2.7)	1.0 (1.2)
pMn $mg\ kg^{-1}$	742 (321)	698 (364)	736 (145)	880 (484)	882 (542)	951 (561)	772 (286)	711 (151)
dFe $\mu g\ l^{-1}$	118 (47)	9 (10)	36 (29)	79 (51)	126 (107)	151 (140)	119 (110)	53 (55)
pFe %	5.9 (1.3)	5.8 (2.0)	6.2 (1.9)	5.5 (1.7)	6.1 (2.3)	5.1 (2.7)	6.1 (2.1)	6.4 (1.9)

Statistical analyses

Temporal changes in total fish catch biomass at each site were estimated using a combination of methods. Linear regressions were performed for each site, with time as the independent variable and biomass (kg) as the dependent variable. In each instance $\ln(x+1)$ transformations were applied to normalise the data. For each site, observed changes in biomass were plotted against time and estimated biomass values, as derived from the linear regression, with upper and lower 95% confidence limits, were then superimposed. Percentage change in biomass at each site was determined from both the observed and predicted data. In the first instance, percentage change was determined by calculating the difference between the observed means for the first and last years over which data were collected. Mean biomass was then recalculated for the same periods, but from the values estimated by the regression equation and the percentage difference again determined.

Differences in fish biomass at each site between years were tested by one-way analysis of variance using quarterly/monthly fish catch data as replicates, with years as classes. Tukey's standardised range tests were applied to identify between year differences where there were significant main effects. Finally, Spearman rank correlations were used to follow monotonic changes in fish biomass over time at each site. The strength and direction of relationships were indicated by the significance level and the correlation coefficient (r; +ve or −ve). All analyses were performed on the longest data series available for each site, with the maximum time period extending from 1983 to 1996. The above approaches were also applied to examine temporal changes in catches of barramundi, *Lates calcarifer*, a dominant riverine species.

Results

Fish catch composition

Since the commencement of biological monitoring, a total of 86 fish species representing 32 families has been captured from sites in the Ok Tedi, Fly and Strickland rivers (Appendix). Catfish in the families Ariidae (16 spp.) and Plotosidae (9 spp.) were the most diverse groups overall. The maximum number of species recorded at any one site ranged from 17 (BIR10) to 60 (FLY10). The ten most abundant species in samples, and their relative proportions in the total catch, are shown in Table 2. Numerically, *Nematolosa* herrings formed the largest proportion of the total catch (37.1%), with the next most abundant species forming 5.6% of the catch. However, in terms of total biomass, *Nematolosa* formed only 4.6% of the total catch, whereas barramundi, *Lates calcarifer*, formed over 30%, and the catfish *Arius leptaspis* over 16%.

Twenty-three species were recorded from riverine sites and not from floodplain habitats (see Appendix). These included six species of ariid catfish, two species of plotosid catfish, three species of mullet (family Mugilidae) and several diadromous species. Conversely, five species were recorded from floodplain habitats and not from any riverine sites (Swales et al. 1999).

The mean number of species recorded from river channel sites (Figure 2) each year since the commencement of monitoring has remained relatively constant (∼20 species). However, in the Ok Tedi the number of recorded species has declined markedly since 1993, with the mean number of species falling from around 15 over the period 1983–1993 to just 3 species in 1996. There was an apparent increase in the number of species in the Ok Tedi in 1993 and 1994, but this was most likely due to the relocation of the Atkamba sampling site (TED 30 to TED35), with the new site being closer to the confluence with the main Fly River and possibly having different habitat conditions (e.g. less river bed aggradation).

Temporal changes in fish catches

Fish catches at most sites showed temporal and spatial variation over the period of sampling. Temporal plots of mean fish biomass and species diversity recorded at each riverine site (Figure 3) show that catches at sites in the Ok Tedi (TED20, TED30), upper Fly (PAR01, FLY02) and middle Fly (FLY10, FLY14, FLY15) have shown a progressive downward trend over the period since the commencement of sampling operations. In addition, catches at sites in the lower Strickland River (STR01) and lower Fly (FLY20) have also declined. However, catches at other sites in the lower Fly (FLY34, FLY36, FLY37) have remained relatively stable over the period of sampling.

Table 2. Total catches of the ten numerically dominant fish species sampled at all riverine sites since the commencement of sampling in the Fly River system up to 30 September 1996.

Species	Total number	% total catch	Total biomass kg	% total catch
Nematolosa spp.	40800	37.11	2526.00	4.610
Melanotaenia splendida	6185	5.63	27.24	0.001
Arius leptaspis	6074	5.52	9018.73	16.450
Thryssa scratchleyi	5757	5.24	771.93	1.410
Lates calcarifer	5391	4.90	16538.75	30.170
Parambassis gulliveri	5336	4.85	489.23	0.890
Ambassis agrammus	4120	3.75	13.75	0.001
Arius berneyi	4070	3.70	2130.24	3.890
Toxotes chatereus	2948	2.68	403.29	0.010
Arius macrorhynchus	2904	2.64	1951.16	3.560

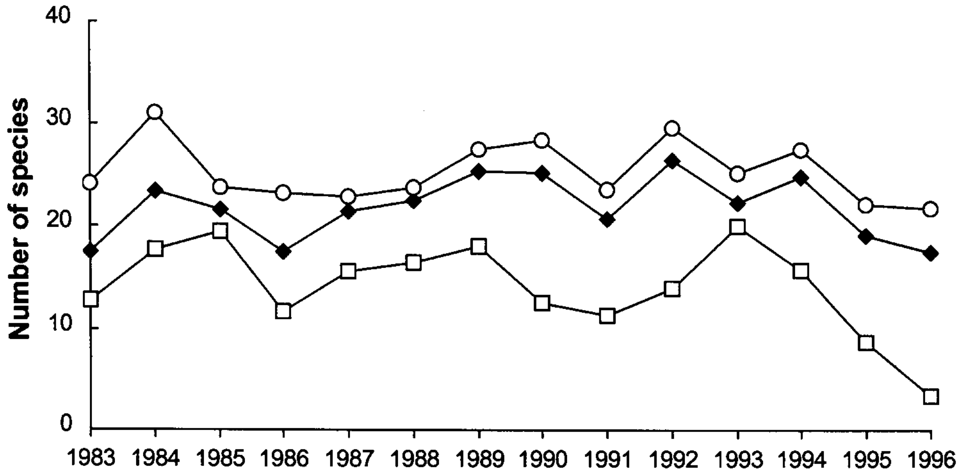

Figure 2. Mean number of fish species recorded at river channel sites since the commencement of sampling (♦ all riverine sites; □ Ok Tedi sites only; ◯ all sites except Ok Tedi).

Mean fish biomass and species richness estimates recorded at each site over the period of sampling are shown in Figure 4. Overall, catches at sites in the Ok Tedi and upper Fly were lower than catches in the middle Fly, lower Fly and Strickland River sites by factors ranging from two to twenty. Fewer species were also recorded at the Ok Tedi and upper Fly sites compared to middle and lower Fly River sites.

Linear regressions of fish biomass with time show significant ($p < 0.05$) reductions in fish biomass at six sites (TED20, TED30, FLY02, FLY10, FLY14, FLY15), with overall estimated reductions in fish biomass, based on predicted data, ranging from 65 to 96% (Figure 5, Table 3). The greatest long-term reductions in fish biomass were recorded at sites in the Ok Tedi at Ningerum (TED20, gill-nets 82.6%, seine-nets 96.2%) and Atkamba (TED30, 86.8%) followed by the middle Fly at Bosset (FLY14, 73.5%), the upper Fly 'control' site (FLY02, 70.2%), the middle Fly at Obo (FLY15, 69.1%) and Kuambit (FLY10, 65.3%).

ANOVA's also demonstrated significant between-years differences in fish biomass at ten sites. However, there was not always a consistent temporal sequence in between-year changes, although the most recent years tended to be the lowest (Table 4). Spearman rank correlations detected nine negative correlations out of 13 sites tested (Table 5). These findings support the results of the regression analyses.

Barramundi

Since sampling first commenced in the early 1980's, barramundi, *Lates calcarifer*, have consistently formed a high proportion of catches at many riverine sites, particularly in the middle Fly. Temporal changes in the abundance of barramundi in catches at sites in the middle and lower Fly (FLY10, FLY14, FLY15, FLY20) and the 'control' site on the Strickland River (STR01) have been variable, with declines over recent years at some sites (FLY10, FLY14, FLY15), but with stable catches or increases at others (FLY20, STR01) (Figure 6).

The proportion of barramundi in the total catch at Kuambit (FLY10) in the middle Fly has varied considerably since the commencement of sampling in 1983, ranging up to 67% and 87% by number and weight, respectively. However, barramundi generally formed a much lower proportion of the total catch at this site in the 1980's than during the early 1990's, although the proportion has declined over the last few years. The numbers and biomass of barramundi in catches at this site have declined overall during the 1990's. Linear regression analyses detected significant changes in the numbers ($p \leq 0.0002$) and biomass ($p = 0.0024$) of barramundi at Kuambit (FLY10) from 1983 to 1996. Spearman rank correlations detected significant temporal reductions in barramundi numbers ($r = -0.48925$, $n = 96$, $p \leq 0.0001$) and biomass ($r = -0.43056$, $n = 96$, $p \leq 0.0001$). ANOVA detected significant between-year differences for both parameters. Similar reductions in barramundi catches during the 1990's were recorded at Bosset (FLY14) and Obo (FLY15). At Bosset, the proportion of barramundi in the total catch has generally been very variable since sampling commenced in 1983 and in recent years has generally not exceeded 15% and 25%, by numbers and biomass

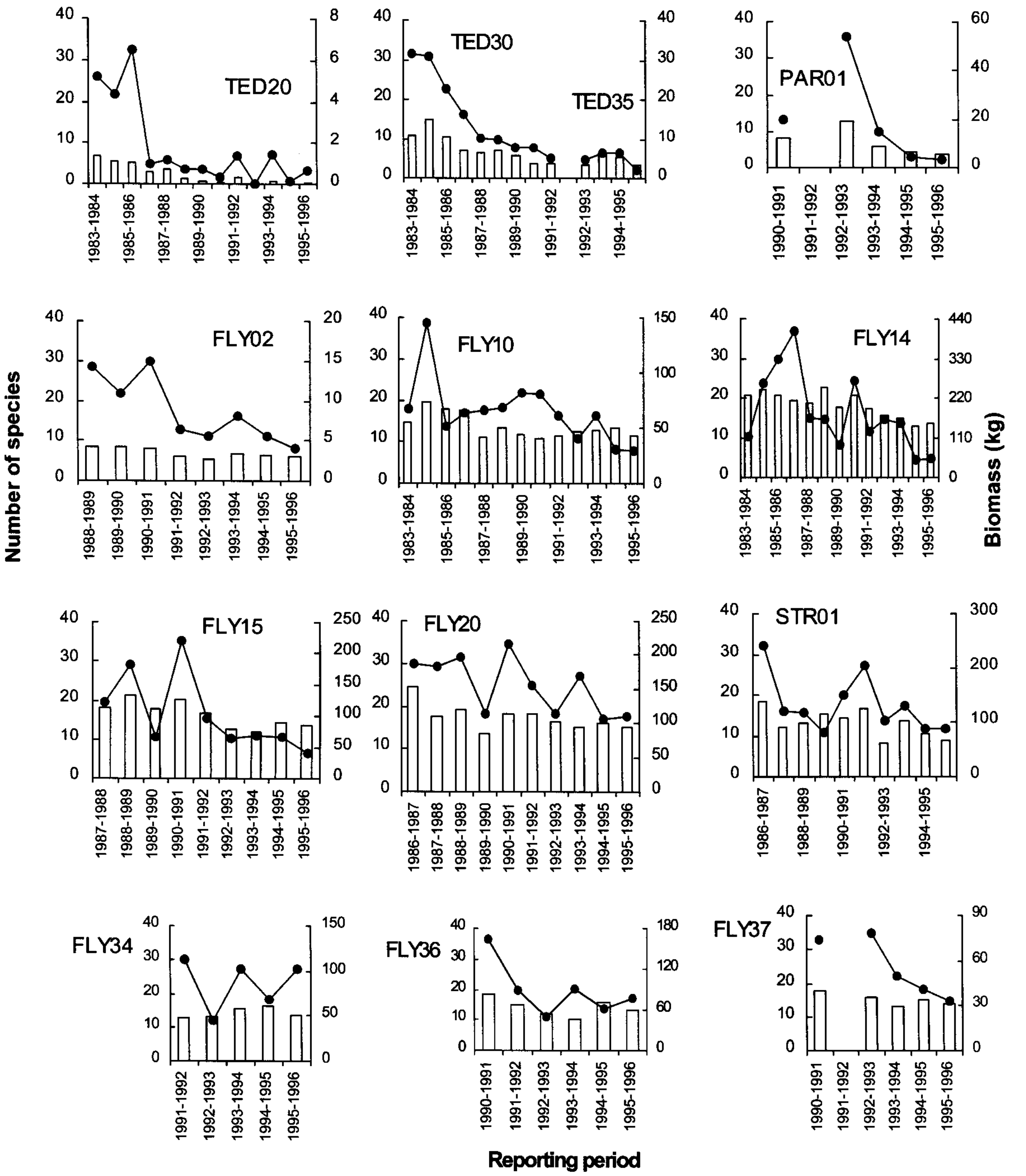

Figure 3. Mean estimates of total biomass (●—●) and diversity (bars) of fish catches at each river channel site since the commencement of monitoring.

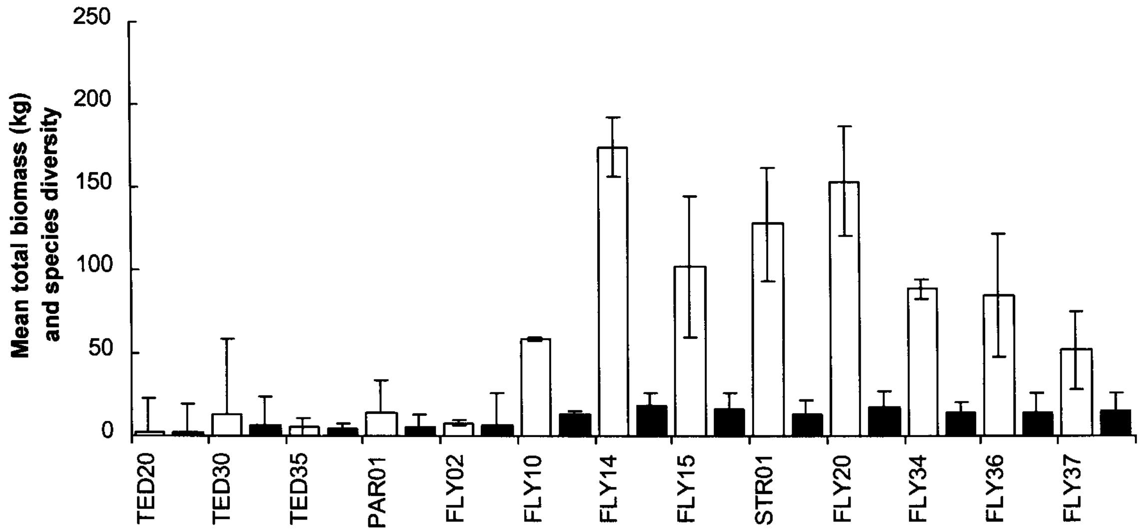

Figure 4. Mean estimates of fish catch biomass (open bars) and species number (shaded bars) (+/−95% confidence limits) at each river channel site since the commencement of monitoring.

respectively. Catch numbers and biomass at this site were high during the 1980's, but have declined during the 1990's. Linear regression analysis revealed significant decreases in numbers ($p \leq 0.0001$) and biomass ($p \leq 0.0001$) at Bosset from 1983 to 1996. Spearman rank correlations detected significant temporal reductions in numbers ($r = -0.65794$, $n = 46$, $p \leq 0.0001$) and biomass ($r = -0.63651$, $n = 46$, $p \leq 0.0001$). ANOVA's also detected significant between-year differences for both parameters (Table 6).

At Obo (FLY15) the proportion of barramundi in the total catch has ranged from 10% to 22%, by number and 20% to 60% by biomass. Catches were high in the 1980's, but have declined in recent years. Linear regression analysis revealed a significant decrease in the number ($p = 0.0305$) and biomass ($p \leq 0.001$) of barramundi over the period from 1987 to 1996. Spearman rank correlation detected significant temporal decreases for catch numbers ($r = -0.4037$, $n = 33$, $p = 0.0198$) and for biomass ($r = -0.529$, $n = 33$, $p = 0.0016$). However, ANOVA's detected no significant between-year differences in numbers and biomass of barramundi at this site (Table 6).

Similarly, linear regression analysis, Spearman rank correlation and ANOVA failed to detect any significant temporal changes in biomass or number of barramundi at Ogwa (FLY20) or the Strickland River site (STR01). Catches at these sites have remained relatively stable over the periods of sampling and there is no evidence of a decreasing trend in catches at either site (Figure 6).

At both Ogwa and the Strickland River site, the proportion of barramundi in the total catches has been high in recent years, ranging up to 40% by number and 75% by biomass.

Discussion

The first systematic survey of the fish fauna of the Fly River, undertaken in 1975 by Roberts (1978), recorded 103 species in 33 families and the study provided the basis for an account of distribution, habitats, food habits, reproduction, migratory behaviour and systematics of the Fly ichthyofauna. It was found that the fauna was characterised (i) by groups which are absent or poorly represented in riverine fish faunas of other regions, particularly ariid and plotosid catfishes, (ii) the unusually large size of many species, (iii) a paucity of very small species, (iv) endemic montane species in Plotosidae and Clupeidae, (v) stenophagous molluscivorous and carcinophagous ariid catfish and (vi) a diverse community of forms suspected or known to have catadromous life-histories. The fact that the present study recorded fewer fish species may be due to a more limited range and number of sampling sites and that the sites in this study were largely restricted to the main river channel and did not include smaller tributaries or floodplain habitats, which are reported elsewhere (Swales et al. 1999).

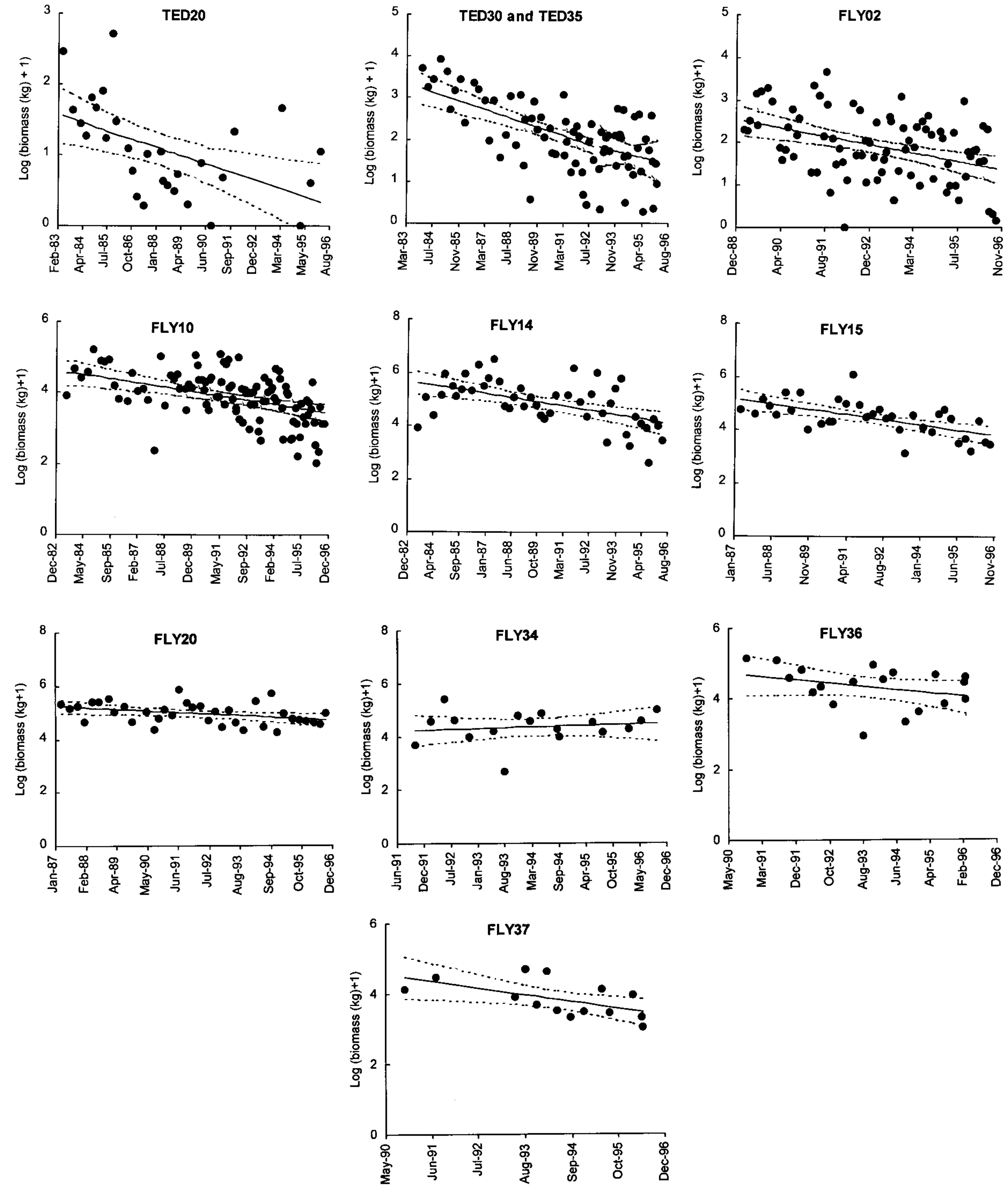

Figure 5. Temporal plots of fish catch biomass ($\log_e(x + 1)$) at each river channel site and estimated linear regression (solid line) with 95% confidence limits (dashed lines).

Table 3. Summary of regression analysis of temporal reductions in fish biomass at each site (ns = regression not significant, non-significant regressions with p values close to 0.05 are indicated, non-significant percent reductions are in parentheses).

Site	Significance level of regression	Sample size	Start of test period	End of test period	Estimated percent reduction in fish biomass
TED20 **gill-net**	0.0032	29	1983/1984	1996	**85.98**
TED20 **seine-net**	0.0001	45	1983/1984	1995/1996	**96.21**
TED30	<0.0001	45	1983/1984	1992	**86.81**
TED35	ns	31	1993	1996	(29.90)
PAR01	ns (0.0991)	11	1991/1993	1995/1996	**89.56**
FLY02	0.0096	81	1989	1996	**70.24**
FLY10	0.0001	103	1983/1984	1996	**65.34**
FLY14	0.0021	46	1983/1984	1995/1996	**73.54**
FLY15	0.0053	35	1987	1995/1996	**69.06**
STR01	ns	35	1987	1996	(31.83)
FLY20	ns	36	1987	1996	(40.25)
FLY34	ns	17	1991/1992	1996	(−22.48)
FLY36	ns	19	1990/1991	1996	(45.76)
FLY37	ns	14	1990/1991	1996	(61.19)

The number of fish species recorded from river channel sites in the Fly River system (86) substantially exceeded the number of species recorded over the same period from sites in floodplain habitats (66), although diversity at the family level was slightly lower (Swales et al. 1999). There was an increasing number of species and catch (biomass and number of individuals) with increasing distance from the headwaters. This relationship correlates with increasing catchment area and area of floodplain, both factors known to have significant effects on catch per unit effort in riverine fisheries in general (Welcomme 1985) and also for the Fly River (Smith et al. 1990). The constancy of the mean number of species recorded at most riverine sites, with the exception of the Ok Tedi, suggests that community diversity is not being adversely affected by mine operations or other environmental disturbances. However, the marked decline in species number recorded in the Ok Tedi over the last four years is also indicative of marked changes in community composition (Swales & Storey unpublished data).

The major environmental impacts arising from the discharges of mine wastes into the headwaters of the system arise from the introduction of large amounts of sediment and copper [80 000 tonnes per day (tpd) of tailings and 120 000 tpd of mined waste rock]. The delivery of mine-derived sediment material to the Ok Tedi has increased since 1984 from 5 million ton per annum (mta) to a current rate of ∼100 mta, of which ∼28 mta is transported through the Ok Tedi to the middle Fly (OTML[1]). The discharge of this waste material has resulted in major physical and chemical changes to fish habitat. The key water quality parameters affected by mine operations are total suspended sediment (TSS), particulate copper (pCu) and dissolved copper (dCu). (Metals such as lead, zinc and cadmium are not elevated above background levels in the total orebody, but are somewhat elevated in skarn ores, which constitute part of the orebody.) The level of each of these parameters varies spatially and temporally. High TSS loads have caused sedimentation of the river bed due to exceedance of the carrying capacity of the flow. This has resulted in river bed aggradation by up to 5 m above baseline in the Ok Tedi and 2–3 m in the middle Fly. Levels of TSS in the Ok Tedi and Fly have increased up to ten fold in the lower Ok Tedi compared to pre-mine conditions (Smith et al. 1990).

In the Ok Tedi and Fly River, levels of both dCu and pCu are elevated above baseline. Currently, the mean level of dCu in the lower Ok Tedi is 15 $\mu g\, l^{-1}$,

[1] OTML. 1998. Hydrology Annual Report 1996–1997. Ok Tedi Mining Ltd. 52 pp.

Table 4. Summary of ANOVAs on between-year differences in fish biomass from riverine sites (year differences arranged in descending order).

Site	df	F-value	p	Tukey's multiple range test	
TED20 **gill-net**	11,19	4.42	0.0032	**85 83 94 84 86 96 91 88 87 89 90 95**	**1992/1993 No Catch**
TED20 **seine-net**	12,32	5.00	0.0001	**86 87 83 88 84 85 95 96 89 91 90 93 94 92**	
TED30	9,35	4.97	0.0002	**83 84 85 86 87 89 88 90 91 92**	
TED35	3,27	0.64	ns	—	
PAR01	4,6	3.20	0.0141	—	
FLY02	7,73	2.92	0.0096	**89 91 90 94 93 92 95 96**	
FLY10	13,89	4.85	0.0001	**84 85 91 90 88 89 86 87 94 83 92 93 96 95**	
FLY14	13,32	3.46	0.0021	**87 86 91 85 84 92 88 89 90 93 83 94 96 95**	
FLY15	9,25	3.61	0.0053	**91 88 89 87 92 90 94 93 96 95**	
STR01	9,25	1.13	ns	—	
FLY20	9,26	1.40	ns	—	
FLY34	5,11	0.89	ns	—	
FLY36	6,12	0.70	ns	—	
FLY37	5,8	0.82	ns	—	

compared to 1 $\mu g\,l^{-1}$ baseline, while pCu is currently 1555 $mg\,kg^{-1}$, compared to 50 $mg\,kg^{-1}$ baseline (Boyden et al. 1978). A wide range of factors can affect the biological availability and toxicity of copper, including metal speciation, complexation and the pH and alkalinity of receiving waters. The Australia and New Zealand Environment and Conservation Council guidelines for the protection of aquatic ecosystems (ANZECC 1995) suggest that the total copper concentration should not exceed 5 $\mu g\,l^{-1}$, while the European Community recommends that for ecosystem protection and for waters of the hardness of the Ok Tedi/Fly River system, the dissolved copper concentration should be less than 6 $\mu g\,l^{-1}$ (Gardiner & Zabel[2]). All of the Ok Tedi/Fly River system fails to satisfy the ANZECC guidelines, while the lower Fly River periodically satisfies the European standard. However, the European standard states that 'higher concentrations of copper may be acceptable where the presence of organic matter could lead to complexation'.

Studies which have been undertaken have demonstrated that essentially total complexation of copper by

[2] Gardiner, T. & T. Zabel. 1989. United Kingdom water quality standards arising from European Community directives – an update. Foundation for Water Research report FR 0041. 109 pp.

Table 5. Summary of Spearman rank correlations of fish biomass against time (start of sampling to present). Correlation coefficients and direction (+, increase; −, decrease) and sample size (n) are presented with level of significance.

Site	Significance	Coefficient	n
TED20^{+}	**	−0.608	29
TED20$^{\#}$	**	−0.661	45
TED30/35	ns	—	31
PAR01	**	−0.809	11
FLY02	**	−0.382	81
FLY10	**	−0.516	103
FLY14	**	−0.507	46
STR01	ns	—	35
FLY20	**	−0.435	36
FLY34	ns	—	17
FLY36	ns	—	19
FLY37	**	−0.684	14

ns, not significant; $^{*}p < 0.05$; $^{**}p < 0.01$; + = gill-net data; # = seine-net data.

organic matter is observed in the Ok Tedi/Fly River system (Stauber & Apte[3]) and a recent study showed no evidence of toxicity of Fly River water to algae in the system (Stauber 1995). Furthermore, a range of bioassay and toxicity studies carried out over recent years has shown no acute toxic effects of mine discharges on fishes and other aquatic life (Smith[4], Smith et al. 1990). Previous studies of the effects of mine waste discharges on the fisheries resource in the Fly River have shown a weak, but statistically significant relationship between increasing pCu concentrations and decreasing fish catches (Smith & Hortle 1991, Smith & Morris 1992). However, recent detailed statistical investigations of possible causative factors which may account for the decline of fish catches in the river were not able to identify clear trends between fish catches and environmental parameters which would allow cause and effect to be established (Fox[5]).

Significant declines in fish catches at sites in the Ok Tedi and middle Fly River over the period of monitoring are suggestive of adverse effects associated with the discharge of mine wastes into the system. The presence of a longitudinal gradient of declines in fish catch, from the Ok Tedi to the lower Fly, also points to an effect which varies with distance from source (increasing dilution of impact). However, significant declines in fish catches were also recorded at two 'control' sites in the upper Fly, an area unaffected by the direct effects of mine waste. Environmental monitoring has shown that the discharge of mine wastes into the Fly River system has caused major changes to river chemistry, hydrology and geomorphology (OTML[6]). It is highly likely that these changes have also produced major effects on river biology, which may account for the declines in fish abundance and diversity recorded in this study. However, the causes of these declines are much harder to identify due to the size and complexity of the system and the lack of understanding of physical, chemical and biological processes.

Analyses of metal levels in tissues from a range of fish species sampled at sites in the Ok Tedi/Fly River and its floodplain have shown levels of Cu, Pb and Cd in flesh, liver and kidney to be elevated at some impacted sites compared to some non-impacted 'control' sites, although findings were often not consistent at all sites (Swales et al. 1998). The results suggest that metal levels in fish tissues are elevated as a direct result of exposure to mine-derived tailings and other waste. However, there is currently no evidence that metal levels recorded in these samples are indicative of lethal or sub-lethal effects of mine wastes on fish survival. High metal levels recorded in kidney and liver are indicative of the excretory and regulatory functions of these organs (Heath 1995). However, fish histopathology studies are currently underway to investigate any possible sub-lethal effects of mine wastes on tissues and organs which could indicate reduced fitness.

Although TSS levels and rates of river bed aggradation have increased substantially over baseline premine levels, waters of the Fly River, and other river systems in New Guinea, are naturally highly turbid due to the high rainfall and land instability, which introduces large amounts of sediment into the river systems. Indeed, the Strickland River, the main tributary

[3] Stauber, J.L & S.C. Apte. 1996. Bioavailability of copper to algae in the Fly River system, Papua New Guinea. CSIRO Investigation Report CET/IR 464R. 78 pp.

[4] Smith, R.E.W. 1997. Review of laboratory based ecotoxicological testing of Ok Tedi Mining Limited wastes. Report to Ok Tedi Mining Ltd., June 1997. 18 pp.

[5] Fox, D.R. 1996. Ok Tedi Mining Limited environmetrics review and biometrical analysis. Report to OTML by CSIRO Mathematical and Information Sciences (Biometrics Unit). 152 pp.

[6] OTML. 1996. Environmental, financial and risk analysis of various dredging and tailings storage schemes to mitigate mining impacts in the Ok Tedi/Fly River system. Company report no. ENV96-08. Report to the State of Papua New Guinea by Ok Tedi Mining Ltd. 195 pp.

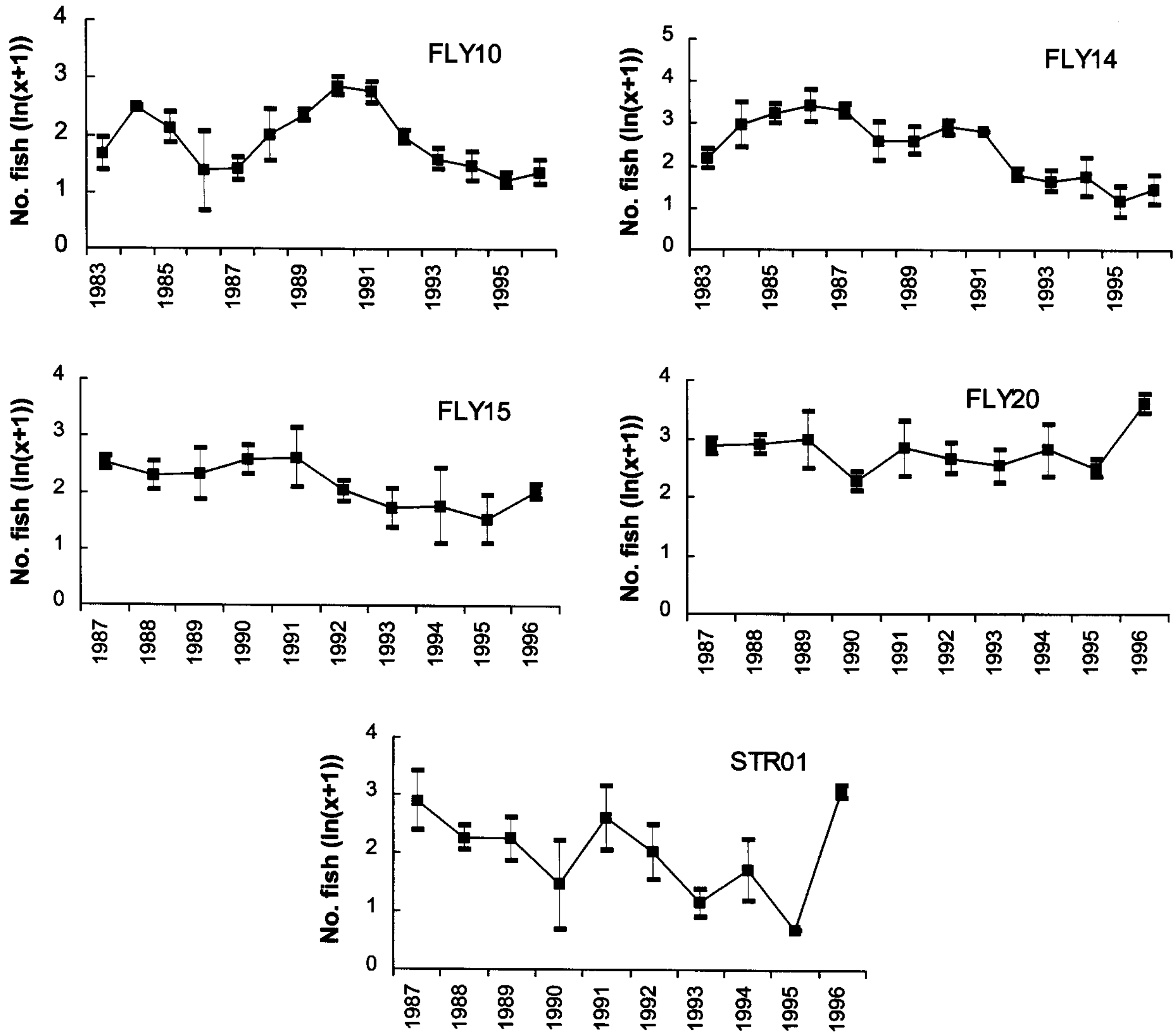

Figure 6. Temporal plots of mean catches (with 95% confidence limits) of barramundi at river channel sites in the middle Fly and Strickland River since the commencement of monitoring.

of the Fly, is naturally more turbid than the Fly, although it now receives small amounts of mine-derived waste material from the Porgera gold mine situated on its headwaters. Previous sampling of the middle Strickland River has shown the fish fauna to be distinctly different from that of the middle Fly River, with the fauna being dominated by large Ariid catfish (OTML[7]). This has been presumed to reflect a fish fauna which is adapted to the higher natural sediment loads in the Strickland River.

In general, the fish fauna of these rivers is much more tolerant of turbid waters than is the fauna of northern hemisphere temperate streams. The fish community in the Fly River has been shown to be very tolerant of high levels of uncontaminated suspended solids (Smith et al. 1990). The presence in the Fly River of large numbers of plotosid and ariid catfish, species which are naturally more tolerant of turbid conditions, appears to lend support to this finding. There is currently no evidence that high levels of TSS per se in the Fly River system are associated with declines in fish catches in the

[7] OTML. 1986. Survey of the fish fauna of the Strickland River at Tiumsinawam with reference to sediment tolerance. Report to Ok Tedi Mining Ltd., ENV86-6. 30 pp.

Table 6. Summary of one-way analysis of variance tests on between-year differences in number and biomass of Barramundi at Fly and Strickland River sites (Tukey's range tests present between-year differences in descending order).

Site	df	F-value	p	Tukey's multiple range test
FLY10				
Numbers	13,82	7.71	<0.0001	**90 91 84 89 85 88 92 83 93 94 87 86 96 95**
Biomass	13,82	5.93	<0.0001	**91 90 89 84 85 92 88 94 93 87 96 83 86 95**
FLY14				
Numbers	13,32	4.86	0.0001	**86 87 85 84 90 91 89 88 83 92 94 93 96 95**
Biomass	13,32	5.44	<0.0001	**87 86 85 91 84 88 90 89 83 92 96 94 93 95**
FLY15				
Numbers	9,23	1.01	ns	
Biomass	9,23	2.19	ns	
FLY20				
Numbers	9,26	1.20	ns	
Biomass	9,26	0.71	ns	
STR01				
Numbers	9,22	2.15	ns	
Biomass	9,22	2.97	0.0181	**87 96 91 89 88 92 94 88 90 93 95**

system. However, it is likely that river bed aggradation, due to mine-derived sedimentation, is associated with declines in fish catches, particularly in the Ok Tedi and upper reaches of the middle Fly. Prior to the discharge of mine waste material, the Ok Tedi had a substrate of predominantly gravel and cobbles, particularly in the higher gradient headwaters, which provided habitat for invertebrate food organisms and a spawning substrate for many fish species. The loss of habitat through sedimentation can reduce the survival of both fish and invertebrate fauna (Cordone & Kelley 1961, Newcombe & Jensen 1996). Although substantial sedimentation has also occurred in the lower Ok Tedi and middle Fly, these low-gradient areas are natural sediment deposition zones and aquatic life may be more adapted to their high sediment levels. Nonetheless, mine-derived sedimentation is known to have modified fish habitat conditions in this region, both in the main river channel and floodplain areas (Higgins 1990). For instance, increased bankside siltation is evident in the Kuambit area. This has lead to the in-filling of backwater areas and smothering of large woody debris, both of which form important habitats for fishes, and may explain the reduced fish catches in this area. Sedimentation of river channel and floodplain areas may be deleterious to fish survival through adverse effects on fish feeding, reproduction and recruitment (Newcombe & Jensen 1996).

The declines in fish catches at sites in the main river channel over the period of monitoring may not necessarily be indicative of overall declines in fish abundance in the system. It is possible that fish in the main channel may be responding to mining-induced declines in habitat and water quality by concentrating in less affected areas, such as tributaries or floodplain areas, where conditions are better than in the main channel. Fish catches at most sites in the extensive floodplain habitats continue to be high and have not shown the same long-term declines as catches in the main river channel (Swales et al. 1999). There is evidence that copper may stimulate locomotory activity in fish and that increased exploratory activity is associated with avoidance behaviour (Heath 1995). Several fish species show an avoidance response to copper in the laboratory at concentrations below the dissolved copper concentrations measured in the Fly River system (Atchison et al. 1987). In the laboratory, Woodward et al. (1997) demonstrated a behavioural avoidance response in cutthroat trout, *Oncoryhnchus clarki lewisi*, when exposed to waters containing a mixture of metals (Cd, Cu, Pb and Zn) or to Cu or Zn alone at levels of $6.0\ \mu g\ l^{-1}$ and $28\ \mu g\ l^{-1}$, respectively. It was concluded that avoidance of metals-contaminated habitats by cutthroat trout may, in part, be responsible for reduced fish stocks. Recent laboratory studies, using fish species native to the Fly River, tested the behavioural response of fishes to waters containing dissolved and particulate copper and mine-derived waste tailings. Preliminary results suggested that some fishes showed an avoidance response to the contaminated water, indicating that they may be able to detect mine-derived pCu (Smith[4]).

In addition to the possible effects of mine-derived waste material, other anthropogenic and natural environmental factors are affecting fishes in the system. The extent of commercial and artisanal fisheries in the Fly River has increased over recent years as villagers have become more affluent and are able to secure more boats, engines and gill-nets. Of the large commercial species, barramundi stocks are most likely to be affected as villagers purchase large mesh gill-nets to target large fish. Also, in recent years, two species of introduced fish, the climbing perch, *Anabas testudineus*, and the walking catfish, *Clarias batrachus*, have become widespread and abundant in areas of the Fly River system, with unknown consequences for native fish stocks (Storey et al. unpublished data).

It is well established that tropical river fisheries show considerable annual and inter-annual variability associated with flooding and other natural climatic factors such as El Niño induced droughts (see Welcomme 1979, 1985, Winemiller 1996, Swales et al. 1999). There is evidence for this from the Fly River. There has been a succession of El Niño droughts in the region in 1982, 1986, 1992–1993 and most recently 1997. During these droughts, the majority of the floodplain, except for deeper oxbow lakes, dries out (see Swales et al. 1999). In response, fish move off the floodplain, away from the shallow waterbodies and into the main river channel. At this time, as fish become concentrated in the main channel, fish catches increase. This occurred in the main channel at Kuambit, Bosset and Obo following the 1992–1993 drought. Catch composition at these times also changed as common floodplain species become more abundant in the main channel. Following the onset of the next wet season and re-flooding of the floodplain, fish catches in the main channel decline as fish disperse back onto the floodplain. The increased dominance of barramundi in catches at riverine sites in 1990–1991 was also thought to be due to natural climatic factors as more fish migrated up the river from coastal spawning areas, following a period of very good recruitment (Hortle[8]).

In conclusion, it is likely that anthropogenic factors associated with the input of mine wastes, and other non mine-related anthropogenic activities, such as commercial and artisanal fishing and introduced fish species, as well as natural climatic events, such as El Niño droughts, are all affecting the fish fauna in the Ok Tedi/Fly River system. However, Ok Tedi Mining Limited is currently taking mitigative measures to reduce the input of mine-derived waste material to the Fly River system which it is hoped will benefit fish and other aquatic life. A number of possible mitigation schemes have been evaluated and a trial scheme has been selected which involves dredging a slot in the lower Ok Tedi to remove mine-derived sediments from the water column. A further possible option, which awaits full evaluation, is the piping of tailings to storage areas in the lower Ok Tedi. It is hoped that the dredging scheme will reduce the rate of river and floodplain sedimentation and so reduce any adverse environmental effects associated with mine operations (OTML[6]).

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[8] Hortle, K.G. 1995. Review of data 1983–94 on catches and condition of barramundi, *Lates calcarifer*. Report to Ok Tedi Mining Ltd. 66 pp.

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Appendix

Fish species recorded from the riverine sampling sites since the commencement of sampling (G = gill net; S = seine net; R = sampled at riverine sites only). Classification according to Allen (1991), Roberts (1978), Munro (1967) and Coates (1993) (*indicates species sampled for tissue metals analysis).

Family	Species	Site													
		BIR10	TED20	TED30	TED35	PAR01	FLY02	FLY10	FLY14	FLY15	STR01	FLY20	FLY34	FLY36	FLY37
Anabantidae	*Anabas testudineus* (climbing perch)				G			G	G	G	G	G	G	G,S	
Apogonidae	*Glossamia aprion** (mouth almighty)	S	G,S	G,S	G		G,S	G,S	G	G					
	Glossamia sandei (sande's mouth almighty)		S												
	Glossamia trifasciata (tree-barred mouth almighty)	S	G,S	S			S	G,S							
Ariidae	*Arius A* (R)										G				
	*Arius B**(R)				G						G	G	G	G,S	G,S
	*Arius augustus** (short barbelled catfish)			G,S			G	G,S	G	G	G	G	G	G	G
	*Arius berneyi** (berney's catfish)			G,S	G	G	G	G,S	G	G	G	G	G	G,S	G
	*Arius carinatus** (comb-spined catfish)			G,S	G			G,S	G	G	G	G	G	G,S	G
	*Arius latirostris** (broad-snout catfish)	S	G,S	G,S	G,S	G	G,S	G,S	G	G	G	G	G		
	*Arius leptaspis** (triangular shield catfish)	S		G,S	G	G	G	G,S	G	G	G	G	G	G,S	G,S
	*Arius macrorhynchus** (sharp-nosed catfish)		G	G,S	G		G	G	G	G	G	G			
	Arius taylori (R)* (Taylor's catfish)		G,S	G	G	G	G	G			G				
	Arius crassilabris (R)* (thick-lipped catfish)			G,S	G	G	G	G	G	G	G	G	G	G	G
	*Cinetodus froggatti** (Froggatt's catfish)			G				G	G	G	G	G	G	G	G
	*Cochlefelis danielsi** (Daniel's catfish)			G,S			G	G	G	G	G	G	G	G,S	G
	*Cochlefelis spatula** (duckbilled catfish)		G	G,S	G,S	G	G	G,S	G	G	G	G	G	G,S	G
	Doiichthys novaeguineae (spoon-snouted catfish)													G	G
	*Nedystoma dayi** Day's (catfish)		G	G,S			G,S	G,S	G	G	G	G	G	G,S	G

	Tetranesodon conorhynchus (Lorentz catfish) (R)										G	G	G	G	G
Atherinidae	*Craterocephalus randi* (Kubuna hardyhead)	S	S	S	S	G	S	S						S	
Belonidae	*Strongylura kreffti** (freshwater longtom)	S		G,S	G	G	G,S	G,S	G	G	G	G		G,S	
Callionymidae	*Callionymus enneactis* (dragonet)													S	
Carcharhinidae	*Carcharhinus leucasi* (bull shark) (R)											G	G	G	G
Centropomidae	*Lates calcarifer** (barramundi)	S		G	G	G	G	G,S	G	G	G	G	G	G,S	G
Chandidae	*Ambassis agrammus* (glass perchlets)	S	S	G,S	G,S		G,S	G,S	G	G	G	G	G	S	G
	Ambassis spp.						S	G							
	*Parambassis gulliveri** (giant glass perchlet)	S	G,S	G,S	G	G	G,S	G,S	G	G	G	G	G	G,S	G,S
Clariidae	*Clarias batrachus* (walking catfish)								G	G	G				
Clupeidae	*Clupeoides papuensis* (toothed river herring)	S	G,S	G,S	S	G	G,S	G,S		G			G	G,S	
	Clupeoides venulosus (West Irian River sprat) (R)							S							
	Nematalosa spp. *(flyensis & papuensis)** (herrings)			G,S	G,S	G	G,S	G,S	G	G	G	G	G	G,S	G
Cynoglossidae	*Cynoglossus heterolepis* (freshwater tongue sole) (R)		G,S	S			S						G	S	G,S
	Cynoglossus spp.													S	G
Datnioididae	*Datnioides quadrifasciatus* (four-banded tigerfish)		G,S	G	G		G	G	G	G	G	G	G	G	G
Eleotrididae	*Mogurnda mogurnda* (trout mogurnda)			S				S		G					
	Mogurnda cingulata (banded mogurnda)		S												
	Ophieleotris aporos (snakehead gudgeon)											G			G
	Oxyeleotris fimbriata (fimbriate gudgeon)			G,S			G	G,S	G	G	G	G		G	G
	Oxyeleotris herwerdini (blackbanded gauvina)				G		G	G	G	G		G		G,S	G
	Oxyeleotris lineolatus (sleepy cod)			G,S	G	G	G,S	G,S	G	G	G	G	G	G	G
	Oxyeleotris nullipora (poreless gudgeon)										G	G	G		
	Oxyeleotris spp.						S	S							

Appendix (*continued*)

Family	Species	Site													
		BIR10	TED20	TED30	TED35	PAR01	FLY02	FLY10	FLY14	FLY15	STR01	FLY20	FLY34	FLY36	FLY37
Engraulidae	*Thryssa rastrosa* (Fly River thryssa)				G		G	G,S	G	G	G	G		G,S	
	*Thryssa scratchleyi** (freshwater anchovy)		G	G,S	G		G	G,S	G	G	G	G	G	G,S	G
Gobiidae	*Glossogobius giurus* (flathead goby)	S	S	S	G		G,S	G,S	G					S	S
	Glossogobius spp.			S										S	
	Glossogobius concavifrons (R) (concave goby)														
	Oxyurichthys papuensi (R)			G,S				G,S							
Hemiramphidae	*Zenarchopterus novaeguinae* (Fly River garfish)	S	G,S	G,S	G,S	G	G,S	G,S				G		G,S	
Kurtidae	*Kurtus gulliveri* (nurseryfish)			G,S			G	G,S	G		G	G	G	G,S	G
Ludjanidae	*Lutjanus argentimaculatus* (mangrove jack)			G				G	G	G	G	G			
	*Lutjanus goldiei** (Papuan black bass)		G		G	G	G	G	G	G	G	G	G	G	G
Megalopidae	*Megalops cyprinoides** (oxeye herring)		G	G	G	G	G	G	G	G	G	G		G	G
Melanotaenidae	*Melanotaenia splendida* (red-striped rainbowfish)	S	G,S	G,S	G,S	G	G,S	G,S	G	G		G			
Mugilidae	*Crenimugil heterocheilus** (R) (fringe-lipped mullet)	S		G,S		G		G,S	G					S	
	Liza alata (basket mullet)			G			G,S	G,S	G	G	G	G	G	G	G
	Liza macrolepis (large-scale mullet)													G	
	Liza parmata (R)					G								S	G
	Liza subviridis (greenback mullet) (R)							G					G		G
Osteoglosidae	*Scleropages jardini** (saratoga)				G		G	G,S	G	G	G	G	G		
Plotosidae	*Neosilurus ater** (narrow-fronted tandan)	S	G,S	G,S	G,S	G	G	G,S	G	G	G	G	G	G	G
	Neosilurus brevidorsalis (short-finned tandan)		G,S												
	Neosilurus equinus (southern tandan)		G,S												

	Neosilurus spp.			G				G							
	Oloplotosus luteus (pale yellow tandan)			G				G,S	G						
	Porochilus meraukensis (Merauke tandan)		S					G,S	G		G			S	
	*Porochilus obesi** (Obbe's tandan)			G,S	G,S		G	G,S	G	G	G	G	G	G	G
	Porochilus spp.								G						
	Plotosus papuensis (Papuan tandan)				G	G	G	G	G			G			G
Pristidae	*Pristis microdon* (sawfish)			G					G	G	G	G	G	G	G
Scatophagidae	*Scatophagus argus* (spotted scat)								G	G		G			
Sciaenidae	*Nibea semifasciata* (sharpnose jewfish)			G	G		G	G	G	G	G	G	G	G,S	G,S
Soleidae	*Aseraggodes klunzingeri* (tailed sole)						S							S	S
Sparidae	*Acanthopagrus berda* (sea bream)			G				G	G	G	G	G	G	G	
Terapontidae	*Amniataba percoides** (tiger grunter)			S	G,S		G,S	G,S	G	G	G				
	Hephaestus fuliginosus (sooty grunter) (R)							G	G						
	Hephaestes roemeri (Roemer's grunter)	S	S	G		G	G	G,S	G						
	Hephaestes raymondi (Raymond's grunter) (R)		S		S			S							
	Pingalla lorentzi (Lorentz's grunter)	S		G,S	G,S			G,S				G			
	Varia lacustris (lake grunter)			S	G		G	G,S	G	G	G	G			
Tetraodontidae	*Chelonodon patoca* (milk-spotted toadfish) (R)												G	S	G,S
	Marilyna meraukensis (Merauke toadfish) (R)												G	S	S
Toxotidae	*Toxotes chatareus** (seven-spot archerfish)			G,S	G	G	G	G,S	G	G	G	G	G	G,S	G,S
	Toxotes lorentzi (Lorentz's archerfish) (R)						S	S	G	G	G	G			
Total number of species		17	27	49	38	23	46	60	50	44	44	47	36	50	41