ASSESSMENT OF HUMAN HEALTH AND ECOLOGICAL RISKS FOR PROPOSED MINE WASTE MITIGATION OPTIONS AT THE OK TEDI MINE, PAPUA NEW GUINEA

SCREENING LEVEL RISK ASSESSMENT

Final Report

Prepared for:

OK TEDI MINING LIMITED
P.O. Box 1
Tabubil, Western Province
Papua New Guinea

Prepared by:

Parametrix, Inc.
5808 Lake Washington Blvd. NE
Kirkland, Washington 98033-7350

URS GREINER WOODWARD CLYDE
4582 South Ulster St.
Suite 1000
Denver, Colorado 80237

November 1999

CONTRIBUTORS

Kevin Brix (Parametrix) – Principal Investigator
Douglas Reagan (URS Greiner Woodward Clyde) – Principal Investigator

David DeForest (Parametrix) – Aquatic Risk Assessment
Jeff Rabbidge (URS Greiner Woodward Clyde) – Terrestrial Vegetation Risk Assessment
Paul Seidel (Parametrix) - Human Health Risk Assessment
Joe Volosin (Parametrix) – Wildlife Risk Assessment

ACKNOWLEDGEMENTS

We are indebted to the following people who helped improve this analysis by furnishing information, advice, and peer review: Simon Apte, Tom Campbell, Don Carroll, Debbie Fetherston, Stephen Flew, Malcolm Lane, Marshall Lee, Monica Rau, Sue Robinson, Eric Schwamberger, Ross Smith, Andrew Storey, Charles Tenakanai, Jat Tinkerame, and Ken Voigt.

We also want to thank the Ok Tedi Peer Review Group for providing detailed reviews and guidance throughout the risk assessment: Peter Chapman (Chair), Margaret Burchett, Peter Campbell, William Dietrich, and Barry Hart.

TABLE OF CONTENTS

TABLE OF CONTENTS (continued)

Page

TABLE OF CONTENTS (continued)

Page

LIST OF FIGURES

LIST OF FIGURES

Figure Page

LIST OF TABLES

ACRONYM LIST

AE	Assessment Endpoint
ANZECC	Australian and New Zealand Environmental Conservation Council
ARD	Acid Rock Drainage
AVS	Acid-Volatile Sulphide
BAF	Biaccumulation Factor
BCF	Bioconcentration Factor
CCME	Canadian Council of Ministers of the Environment
CEC	Cation Exchange Capacity
CSIRO	Commonwealth Scientific Industrial Research Organisation
DLRA	Detailed Level Risk Assessment
DMAA	Dimethyl Arsenic Acid
DO	Dissolved Oxygen
DOM	Dissolved Organic Matter
EC_{50}	Effects Concentration
EED	Expected Environmental Dose
ENSO	El Niño Southern Oscillation
ERL	Effects Range Low
ERM	Effects Range Median
GAE	General Assessment Endpoint
HERA	Human and Ecological Risk Assessment
HQ	Hazard Quotient
LOAEL	Lowest Observed Adverse Effects Level
LOEC	Lowest Observable Effect Concentration
MMAA	Monomethyl Arsenic Acid
MWM	Mine Waste Mitigation
MWMRA	Mine Waste Mitigation Risk Assessment
NOAEL	No Observable Adverse Effects Level
NOEC	No Observable Effect Concentration
ORWB	Off River Water Body
OTML	Ok Tedi Mining Limited
PEL	Probable Effect Level
PNG	Papua New Guinea
RfD	Reference Dose
SEM	Simultaneously Extracted Metals
SLRA	Screening Level Risk Assessment
SOPC	Stressors Of Potential Concern
TDI	Tolerable Daily Intake
TEL	Threshold Effect Level
TSS	Total Suspended Solids
U.S. EPA	United States Environmental Protection Agency
WHO	World Health Organisation

EXECUTIVE SUMMARY

The overall purpose of the Screening Level Risk Assessment (SLRA) was to screen for potential risk posed by a wide range of mine-related chemical and physical stressors (e.g., copper, riverbed aggradation) to various receptors (people, fish, forest). The intent was to "screen out" stressors and receptors that do not require further evaluation and to define the probable boundaries of the area of potential risk. The process included multiple conservative assumptions to ensure that those stressors and receptors "screened out" truly warranted no further consideration. The receptors and stressors not screened out were identified for further evaluation in the Detailed Level Risk Assessment (DLRA), the objective of which is to estimate the relative risks associated with five mine waste mitigation (MWM) options. However, as discussed in the conclusion section of this summary and in the DLRA Executive Summary, not all of the stressors identified in the SLRA were actually considered in the DLRA due to data and time constraints.

The risk evaluation followed a methodology equivalent to those used internationally in Australia, Canada, the European Union and the United States. The SLRA was divided into four sections – Problem Formulation, Exposure Characterisation, Effects Characterisation, and Risk Characterisation. Problem Formulation describes the sources of stressors, the potential receptors and develops conceptual models on how they might interact. Exposure Characterisation quantifies receptor exposure to stressors, while Effects Characterisation describes the potential effects of the stressors on receptors. Risk Characterisation integrates all of the above to describe/quantify potential risk in the system. Each of these sections addressed risk to four different receptor groups – human health, terrestrial vegetation, aquatic life, and wildlife.

The remainder of this Executive Summary is organised in a similar manner describing the SLRA process and results, concluding with a summary of the mine-related stressors, assessment endpoints/receptors, and areas identified for further assessment in the DLRA. The DLRA then quantifies and compares risks between the different proposed mine waste mitigation options.

PROBLEM FORMULATION

The Problem Formulation characterised human and ecological resources of the potentially affected area (Ok Tedi and Fly River basins, and Fly River estuary) and identified potential chemical and physical stressors on humans, terrestrial vegetation, aquatic life, and wildlife. Human populations evaluated consist of the local peoples living in over 100 villages along the Ok Tedi/Fly River system. Terrestrial vegetation types included lower montane rain forest in the vicinity of the mine, foothill rain forest on steep slopes below 1,000 m elevation, lowland alluvial floodplain forest, and a variety of swamp and grassland vegetation types in the floodplain of the middle and lower Fly River. Aquatic habitats included riverine and off river water bodies (ORWBs). Wildlife included mammals (e.g., cuscus and tree kangaroos, frugivorous and insectivorous bats), birds (e.g., cassowary, water and shorebirds, birds of paradise), reptiles and amphibians (e.g., frogs, pythons, goannas), and a diverse assemblage of invertebrates. Aquatic fauna include zooplankton, various macroinvertebrates (e.g., freshwater prawns), turtles, crocodiles, and over 60 species of fish.

Potential risks to humans were evaluated for both children and adults. Potential risks to terrestrial vegetation and aquatic life were assessed assuming that generic biological communities typical of the ecoregion occur in the Ok Tedi/Fly River system (i.e., forest and wetland vegetation). For screening purposes, toxicity data for the most sensitive species of terrestrial vegetation evaluated in the world literature were used. Potential risks to wildlife were assessed using representatives of key species within the food web from either an ecological (e.g., food chain) or cultural perspective (e.g., use in ceremonies). The representative species selected for this assessment were the fruit bat, earthworm, great egret, white-headed stilt, herbivorous turtle, scavenging turtle, estuarine and freshwater crocodile, rusa deer, wild pig, and the cassowary.

A number of chemical, physical, and biological stressors associated with mine waste disposal practices were identified for evaluation in the SLRA. Stressors that could be linked either directly (e.g., copper, riverbed aggradation) or indirectly (e.g., forest dieback as a result of riverbed aggradation) to mine waste discharge were evaluated. The chemical stressors (Table 1) differed slightly for the different receptor groups due to differences in exposure pathways and sensitivity to the stressors.

Table E-1. Chemical stressors of potential concern.

Aquatic	Terrestrial	Human Health
Aluminium	Aluminium	Aluminium
Arsenic	Arsenic	Arsenic
Cadmium	Cadmium	Cadmium
Chromium	Chromium	Chromium
Copper	Copper	Copper
Iron	Iron	Iron
Lead	Lead	Lead
Manganese	Manganese	Manganese
Mercury	Mercury	Mercury
Molybdenum	Molybdenum	Nickel
Nickel	Nickel	Selenium
pH	Selenium	Silver
Selenium	Silver	Zinc
Silver	Zinc	
Zinc		
Mill Reagents		

Physical stressors to terrestrial vegetation identified for evaluation during this phase included: floodplain aggradation, flooding frequency, and scour. The physical stressors identified for evaluation in the aquatic ecosystem included: total suspended solids (TSS); riverbed aggradation; and hypoxia (low dissolved oxygen). The biological stressors considered included potential loss of forest, wildlife and aquatic species as a result of chemical and physical stressors. Potential loss of these resources creates a secondary biological stressor. For example, if it were predicted that fish were at significant risk in the system, fish-eating birds would also be at risk due to impacts on their food resources.

Only mine-related stressors were considered in this assessment because the overall objective of the Human and Ecological Risk Assessment (HERA) is to compare risks between the five proposed MWM options. It is recognised that a wide range of additional stressors (e.g., El Niño, fire, increased fishing pressures, disease, malnutrition) are interacting in the system, but these are considered beyond the scope of the HERA to assess.

EXPOSURE CHARACTERISATION

Human exposures to mine-related chemical stressors were assumed to occur by contact with sediments, surface water, and consumption of locally obtained foods (plants, aquatic live, and wildlife). The amount of chemicals taken into the body (i.e., administered doses) from these exposures were estimated based on concentrations of chemicals in, and estimated contact rates with, each environmental medium, as well as assumptions regarding body weights. Long-term (i.e., chronic) exposure was assessed because the effects from these exposures typically occur at the lowest doses.

There was a unique mixture of terrestrial plant exposure to physical and chemical stressors in different reaches of the Ok Tedi/Fly River systems. The SLRA identified limited exposure of terrestrial plants to chemical stressors in the Ok Mani. However, the forest adjacent to the channel has been physically buried by mine waste and forest vegetation on steeper slopes has been eliminated by scour in some areas. The risk assessment focused on actual and potential exposure of vegetation to physical and chemical stressors in the Ok Tedi and middle Fly River, and potential exposure to elevated concentrations of chemical stressors in the lower Fly River. Due to estuarine dilution effects and increased flood tolerance of the vegetation (i.e., mangroves), no potential exposures from mine-related stressors (chemical or physical) were identified for the Fly River estuary. However, because hydrologic modelling does not address areas downstream of the confluence with the Strickland River (Everill Junction), potential effects below the Strickland cannot be conclusively eliminated.

As data permitted, potential risks to aquatic life were assessed by evaluating chemical stressors in both surface water and sediment. The Ok Tedi/Fly River system was divided into six river reaches, and the estuary. Chemical stressors were screened by pooling data for each reach. Acute exposure to each stressor was expressed as the 95th percentile of the concentration data, while chronic exposure was expressed as the 95 percent upper confidence limit of the mean. Given the high within-reach variability in TSS concentrations, TSS was evaluated on a site-by-site basis.

The wildlife risk assessment also divided the Ok Tedi/Fly River system into separate reaches. Each receptor or receptor group was expected to inhabit a particular area based on the food they consume and their habitat requirements. Doses were estimated for each model receptor based on their assumed exposures to metals measured in surface waters, tissue, soil, and sediment. However, not all wildlife receptors were assumed to be exposed to metals in all four media, and different receptors were exposed to metals in different food sources. For example, the white-headed stilt (*Himantopus leucocephalus*) is the only model receptor expected to consume invertebrates and to incidentally ingest sediment. To estimate exposure, doses were calculated using assumptions on body weights and water, food, and sediment consumption rates as applicable. All dose assumptions (body weight, consumption rate) were derived from the scientific literature.

EFFECTS CHARACTERISATION

The Effects Characterisation described the effects of physical and chemical stressors and developed a series of effects and toxicity thresholds for different stressors against which to compare predicted exposure. For the majority of chemical stressors, previously established and internationally accepted guidelines (World Health Organisation, U.S. Environmental Protection Agency, Australian and New Zealand Environment and Conservation Council, and Canadian Council of Ministers of the Environment) were used. For physical stressors (e.g., TSS, aggradation, flooding frequency), thresholds were developed because none were available in the literature. Methods specific to the four main receptor groups (humans, terrestrial vegetation, aquatic life, and wildlife) are summarised below.

Human Health

Potential human health risks were assessed by comparing estimated doses from contact with environmental media to internationally accepted toxicity thresholds (non-carcinogens) or to an estimate of the carcinogenic potency factor for carcinogens. Arsenic was the only chemical evaluated that has the potential to cause cancer.

For non-carcinogenic stressors, the risk was characterised by dividing the estimated dose by the toxicity threshold to calculate a hazard quotient. When the hazard quotient exceeds one, there is a potential for risk and further evaluation is generally recommended. For carcinogens (i.e., arsenic), the estimated dose was multiplied by a cancer potency factor to determine the probability of incremental excess cancer risk in the exposed population. An excess risk of less than one in a million is generally considered acceptable although naturally occurring levels of arsenic can result in risk predictions in excess of one in a million and in some cases approaching one in ten thousand.

Terrestrial Vegetation

The effects of physical stressors on terrestrial vegetation vary, depending on the severity of the stress and duration of exposure. Effects of scour are loss of all vegetation in affected areas. Effects of floodplain aggradation and flooding frequency produced similar effects including inhibition of soil respiration, chlorosis, leaf loss, root death, and eventual plant death. While some species variability has been observed, the general effect of prolonged exposure is the death of all plants comprising a particular vegetation community in a given location and their eventual replacement with more flood-tolerant forms.

Threshold effects levels for flooding frequency were set at ≥30 percent annual inundation for forest areas and ≥60 percent for grassland wetlands. For floodplain aggradation, sediment depths ≥30 cm were used. While these values were later refined in the DLRA, they were sufficiently conservative to screen in both stressors and all potentially affected river reaches.

The effects of chemical stressors on terrestrial vegetation include chlorosis, reduced growth, poor root development, and plant death. Effects threshold levels for each chemical stressor were derived from the literature using current standard international sources. Uncertainties in applying these thresholds included differences in sensitivity between receptor species in the area from those investigated in the literature and bioavailability of these stressors to exposed plants. Concentrations of these chemicals in floodplain sediments/soils were compared to either effects threshold levels or background to screen in contaminants.

Aquatic Life

Screening criteria for aquatic life used for comparison with the stressor exposure concentrations included surface water and sediment guidelines from various sources (e.g., Papua New Guinea, Australia/New Zealand, Canada, and the United States). All of these sources provide chronic surface water guidelines, while acute guidelines were only available from the U.S. The surface water guidelines used are generally designed to protect 95 percent, or all of, a generic aquatic community. The sediment guidelines were derived based on a large database of sediment toxicity results from both field and laboratory studies. These guidelines tend to be conservative. For example, guidelines for aluminium, chromium, copper, nickel, and zinc were less than background concentrations for the Ok Tedi/Fly River system as determined in adjacent watersheds that were unaffected by mining activities. For these metals, sediment concentrations were screened against background concentrations. Screening criteria for physical stressors included comparison to pre-mine concentrations and conservative thresholds derived from the literature. No data were available for deriving a threshold for riverbed aggradation, so a threshold of 1 metre was set arbitrarily.

Wildlife

To evaluate a chemical's toxicity to wildlife receptors, chronic toxicological effects data were obtained from the scientific literature. The effects data were generally based on adverse effects to reproduction, growth, and development. The lowest most protective toxicity data were used in this assessment. In all cases chronic values were unavailable for the receptor to be evaluated, therefore, surrogate species (e.g., rat, chicken, quail) were used. If toxicological effects data for a bird were not available, data for mammals were used. Similarly, because little or no toxicity testing has been performed on reptiles, data for birds were used as a surrogate for reptiles. Because of the uncertainty associated with these extrapolations, additional safety factors were applied to the effects thresholds. The indirect effects of physical stressors on wildlife (e.g., loss of forest habitat) were only assessed qualitatively, so no thresholds were derived.

RISK CHARACTERISATION

The risk characterisation integrated information from the exposure and effects characterisations to provide estimates of risk. In the SLRA, this was most frequently accomplished by use of the hazard quotient (HQ). The hazard quotient is defined as:

$$HQ = \frac{\text{Expected Environmental Concentration}}{\text{Toxicity Threshold}}$$

Hazard quotients >1 for a given stressor and receptor are generally considered high enough to warrant further investigation in a DLRA. However, the relationship between the hazard quotient and risk is not linear. In other words, an HQ of 10 is not necessarily 10 times worse than an HQ of 1, but it does imply potentially greater risk and therefore a greater need for evaluation in a DLRA. The other important point regarding hazard quotients is that they are not all equal and are influenced by the assumptions on which they are based. For example, human health risk assessments are generally more conservative than aquatic life risk assessments. Consequently, HQs of 1 or 2 are commonly considered insignificant in human health risk assessments (depending on the stressor), but potentially significant in aquatic life risk assessments. The use of best professional judgement regarding the significance of the HQ is just as important as the absolute value in determining whether further assessment of a stressor is required in a DLRA. This is reflected in the risk characterisation described below.

<u>Human Health</u>

Generally, negligible risks to human health were predicted from most chemicals and exposure pathways. However, a few exceptions are discussed below. Exposure to chemicals in food generally resulted in the highest intakes. Hazard quotients for cadmium in the lower Ok Tedi and Fly River estuary exceeded one based on the consumption of fish and shellfish. Intakes of cadmium exceeded levels believed to be safe for consumption over many years (i.e., 45 to 50). Cadmium risks predicted in the lower Ok Tedi exceeded those from reference areas, while those in the estuary were of a similar magnitude as the reference areas, suggesting that elevated cadmium concentrations reflect naturally occurring conditions. The potential risk estimates for cadmium were based on the assumption of long-term consumption because effects (i.e., an increase in protein or enzymes excreted in urine) will only occur after many years of exposure to elevated concentrations. If levels of cadmium in fish tissue decline in the future, the actual risks would be lower than those predicted. Based on the available information, the potential risk from cadmium does not warrant additional investigation in the DLRA, but does warrant continued monitoring to ensure that exposure does not increase over time.

Intakes of lead also exceeded levels believed to be safe in the estuary, although the magnitude of the exceedance was low (HQ =1.7). Similar to cadmium, lead is known to accumulate in shellfish. Thus, the extent to which this risk prediction reflects actual risks is partially dependent on the extent to which shellfish are used in the diet, which has not been well characterised. Further, blood lead data were not available to validate whether exposures from lead, regardless of whether they are mine-related, may be occurring in the Fly River estuary or reference area villages. Similar to cadmium, the information available indicates lead should continue to be monitored but does not warrant further investigation in a DLRA.

Chemical concentrations in human hair were available providing an additional line of evidence concerning levels of exposure to metals. Although these data represent exposures from all sources and cannot be tied to mine-related exposures, they do allow some assessment of exposure to mercury for which no other environmental data were available in the SLRA. These data reveal that, in general, people living along the lower Fly River have higher levels of hair mercury than those on the Ok Tedi. This result is not unexpected and is consistent with literature reports of higher hair mercury among populations that eat more fish. Additionally, net methylation rates are likely greater on the lower Fly River than on the Ok Tedi due to the greater prevalence of wetlands. Thus, the higher hair mercury concentrations on the lower Fly River are likely due to some combination of both of these factors. Although some individuals show hair concentrations that may be approaching levels of concern, most people do not appear to be exposed at levels of concern and hair concentrations overall are generally within or below levels that are typical for fish-eating populations worldwide.

The potential for adverse effects to aquatic or terrestrial resources used by local people was evaluated in the aquatic, wildlife, and terrestrial vegetation components of the SLRA. Potential risks to all of these components were predicted. Consequently, there are potential adverse indirect effects to local peoples through loss of these resources. These risks were further evaluated in a qualitative manner in the DLRA.

Terrestrial Vegetation

Physical stressors to terrestrial vegetation (floodplain aggradation, flooding frequency, and scour) were screened by comparing known levels of these physical factors with conservative effects thresholds. Based on threshold criteria compared to field data and hydrologic modelling predictions, the Ok Tedi and middle Fly River had exceedances for flooding frequency and/or floodplain aggradation. Therefore, all physical stressors were retained for further evaluation in the DLRA. Scour effects were noted for the Ok Mani, lower Ok Tedi, and upper middle Fly River. Observed or predicted levels of physical stressors did not exceed screening thresholds for the Fly River below the confluence with the Strickland River. However, due to high uncertainties and limited data, effects below this point are possible. Field evidence of risks to vegetation came from recent aerial photography and OTML investigations on vegetation dieback.

A hazard quotient approach was used to screen for chemical stressors to terrestrial vegetation. A range of chemical stressors were identified in mine-derived sediments along the riverbank in the upper Ok Tedi at concentrations in excess of the respective toxicity-based soil/floodplain sediment thresholds and estimated background concentrations.

Mine-derived sediments along riverbanks of the Ok Mani and upper Ok Tedi may contain chemical stressors at concentrations that pose risk to riparian vegetation; however, terrestrial vegetation away from the riverbank is not likely to be exposed to these stressors. Copper, lead, molybdenum, and silver are present in floodplain sediments throughout the lower Ok Tedi and middle Fly River at concentrations that could pose unacceptable risk to terrestrial vegetation. No pertinent floodplain sediment data were available from the lower Fly River or estuary. Analytes and their potential to pose unacceptable risk to terrestrial plant communities are listed in Table 2.

Table E-2. Analytes and their potential to pose an unacceptable risk to terrestrial plant communities.

Analyte	Ok Mani[1]	Upper Ok Tedi	Lower Ok Tedi[2]	Upper Middle Fly River	Lower Middle Fly River
Aluminium	No	No	No	No	No
Arsenic	No	Yes	Yes	Yes	Yes
Cadmium	No	Yes	Yes	No	No
Chromium	No	No	No	No	No
Copper	No	Yes	Yes	Yes	Yes
Iron	No	NG	--	--	NG
Lead	No	Yes	Yes	Yes	Yes
Manganese	No	Yes	Yes	Yes	Yes
Molybdenum	No	Yes	Yes	Yes	Yes
Nickel	No	No	No	No	No
Selenium	No	--	--	--	--
Silver	No	Yes	Yes	Yes	Yes
Zinc	No	Yes	Yes	Yes	Yes

1 Low potential for exposure due to the highriver flows and limited potential for sediment depostion
2 Potential for risk estimated from upper Ok Tedi and upper middle Fly River sediment data.
Yes Analyte may potentially pose an unacceptable risk to terrestrial vegetation communities at the particular reach.
No Analyte is unlikely to pose an unacceptable risk to terrestrial vegetation communities at the particular reach.
-- No sediment data available. NG = No phytotoxicity-based guideline or benchmark available.

Aquatic Life

Potential stressors for aquatic life were screened by comparing concentrations to conservative effects levels. If the exposure concentration exceeded the effects concentration, the stressor was identified as a stressor of potential concern (SOPC). The SLRA eliminated some chemicals in some locations from further evaluation; the remaining stressors identified as being of potential concern are shown in Table E-3. Hypoxia was not identified as an SOPC for further evaluation in the DLRA, but additional monitoring of dissolved oxygen in the system was recommended.

Wildlife

For wildlife, hazard quotients were calculated by dividing the estimated dose (or direct exposure concentration in the case of earthworms) by the wildlife threshold concentration for each chemical stressor. None of the HQs for the cassowary, great egret, estuarine crocodile, freshwater crocodile, herbivorous turtle, scavenging turtle, rusa deer, or wild pig exceeded one, thereby suggesting negligible risk to these receptors from chemical stressors. Hazard quotients for the fruit bat, white-headed stilt, and terrestrial invertebrates, however, exceeded one, indicating potential risk to these receptors (Tables E-4 and E-5).

Table E-3. Surface water and sediment SOPCs for aquatic life.

		Surface Water		
Location	Site-Type	Acute	Chronic	Sediment
Ok Mani	Main Stem	None	TSS	Aggradation
Upper Ok Tedi	Main Stem	Cu, TSS	Cu, Se, TSS	Ag, As, Cd, Cu, Pb, Mn, Ni, Zn, aggradation
Lower Ok Tedi	Main Stem	Cu, TSS	Cd, TSS	Aggradation, No metals data available.
	ORWB	---	---	---
	Floodplain	---	---	---
Upper Middle Fly River	Main Stem	Cu, TSS	TSS	As, Cd, Cu, Pb, Mn, Zn, Aggradation
	ORWB	Cu	Cu	Cu
	Floodplain	---	---	Ag, Cu, Pb, Zn
Lower Middle Fly River	Main Stem	Cu, TSS	Fe, Pb, TSS	Cd, Cu, Pb, Zn, aggradation
	ORWB	None	Fe, TSS	Al, Cr, Cu, Ni, Pb, Zn
	Floodplain	None	Fe, TSS	Cu
Lower Fly River	Main Stem	TSS	TSS	None
	ORWB	---	---	---
	Floodplain	---	---	---
Estuary		---	---	---

--- = No data

Potential risk to the white-headed stilt posed by copper, lead, and zinc was primarily from incidental ingestion of sediment (Table E-4). There were also risks predicted in the lower middle Fly River for the white-headed stilt exposed to aluminium. However, risk from exposure to aluminium at background sites was higher. Exposures to aquatic invertebrates were based on estimates using a bioconcentration factor from the literature and may not be appropriate for site-specific conditions.

Potential risks to the fruit bat were from exposure to arsenic, copper, iron, and lead via food ingestion in the upper Ok Tedi (Table E-5). Potential risks to the fruit bat also were predicted from exposure to arsenic via food in the upper middle Fly River and copper exposure via ingestion of food in the upper Ok Tedi and middle Fly River. Risks also were predicted for aluminium in these same reaches. However, risks for aluminium were higher at background locations and so aluminium was not considered of concern. These risk predictions were based on a number of assumptions including the use of main stem or floodplain sediment data as a surrogate for soil data and use of bioaccumulation factors to predict metal concentrations in fruit.

Table E-4. Wildlife HQs for aquatic pathways.

Stressor	Location	Site-type	HQ	Receptor	Primary Source of Exposure
Copper	Upper Middle Fly River	ORWB	1.0	Stilt	Incidental sediment ingestion
Lead	Lower Middle Fly River	ORWB	3.2	Stilt	Incidental sediment ingestion
Zinc	Lower Middle Fly River	ORWB	2.3	Stilt	Incidental sediment ingestion

Table E-5. Wildlife HQs for terrestrial pathways.

Stressor	Location	Site-type	HQ	Receptor	Primary Source of Exposure
Copper	Upper Ok Tedi	Main stem	47	Terrestrial invertebrate	Direct soil contact
	Upper Middle Fly River	Flood plain	9.1	Terrestrial invertebrate	Direct soil contact
	Upper Middle Fly River	Flood plain	4.0	Terrestrial invertebrate	Direct soil contact
Iron	Upper Ok Tedi	Main stem	544	Terrestrial invertebrate	Direct soil contact
Lead	Upper Ok Tedi	Main stem	3.2	Terrestrial invertebrate	Direct soil contact
Manganese	Upper Ok Tedi	Main stem	9.8	Terrestrial invertebrate	Direct soil contact
Zinc	Upper Ok Tedi	Main stem	94	Terrestrial invertebrate	Direct soil contact
	Lower Middle Fly River	Main stem	16	Terrestrial invertebrate	Direct soil contact
Arsenic	Upper Ok Tedi	Main stem	12	Fruit bat	Food ingestion
	Upper Middle Fly River	Flood plain	4.3	Fruit bat	Food ingestion
Copper	Upper Ok Tedi	Main stem	17	Fruit bat	Food ingestion
	Upper Middle Fly River	Flood plain	3.3	Fruit bat	Food ingestion
	Lower Middle Fly River	Flood plain	1.5	Fruit bat	Food ingestion
Iron	Upper Ok Tedi	Main stem	29	Fruit bat	Food ingestion
Lead	Upper Ok Tedi	Main stem	6.7	Fruit bat	Food ingestion

Finally, risks from copper, iron, lead, manganese and zinc were predicted for terrestrial invertebrates via direct contact with soil in the upper Ok Tedi, from zinc in the upper middle Fly River, and from copper in the upper and lower middle Fly River (Table E-5). As with risks for other receptors exposed to soil, floodplain sediment data were used as a surrogate for soil data. Although risks were also predicted for aluminium and chromium, hazard quotients for background sites were higher. Therefore, aluminium and chromium were not considered as being of concern for terrestrial invertebrates. The magnitude of some of the HQs (e.g., 544 for iron in the upper Ok Tedi) in the SLRA suggests some uncertainty in regard to risk for terrestrial invertebrates. These uncertainties exist, in part, due to the conservative assumptions used in this SLRA (e.g., terrestrial invertebrates are exposed to soils with concentrations similar to those in sediment and metals in soil are 100 percent bioavailable). It is recommended that iron be further evaluated in a DLRA.

UNCERTAINTIES

There are always uncertainties associated with conducting a risk assessment. Part of the risk assessment process is to attempt to quantify or at least describe the uncertainties. The majority of the uncertainties were due to either assumptions made in the risk assessment process or to data gaps. The more significant uncertainties associated with this SLRA are summarised below.

Human Health

- Children are potentially more sensitive to chemical stressors than adults. The exposure characterisation and effects characterisation were based on a lifetime exposure integrating risks to both children and adults. A secondary assessment was performed to assess potential risks to children independently. This assessment introduced additional conservatism because the toxicity thresholds assume a lifetime (40-70 years) exposure. Using this additional level of conservatism, several additional stressors (chromium, iron, manganese, and nickel) had HQs >1. However, all HQs were <4 and most were <2 using this approach. Considering the additional conservatism in this evaluation and the low HQs that were calculated, there do not appear to be any unique risks to children.

- The rate of consumption of aquatic resources (i.e., fish and shellfish) and other foods (e.g., sago and cassava) from affected rivers, the Fly River estuary, or other mine-affected and reference area harvest locations was not well characterised. Further, the extent to which fish species represented in the chemistry database compare to those eaten by local people in the Ok Tedi or Fly Rivers and estuary was not known;

- Although mercury levels in hair do not suggest excessive exposure, no definitive assessment of risks from mercury was performed because mercury data were unavailable for food items to corroborate the hair data results;

- Data in fish tissues were limited to four stressors – cadmium, copper, lead and zinc; and

- Assumptions were made regarding exposures largely based on human use and other data from the Strickland River catchment. The applicability of these data to the Ok Tedi and Fly River are uncertain.

Terrestrial Vegetation

- No site-specific data were available on phytotoxicity of the potential chemical stressors;

- Limited data were available for chemical stressor concentrations in floodplain sediments. Floodplain sediment data were available for most chemical stressors for the lower Ok Tedi and upper middle Fly River. Only copper data were available for the lower middle Fly River. No floodplain sediment data were available for the lower Fly River ;

- Because of the diversity in flooding and aggradation tolerance levels among plant species within the potentially affected areas, effects thresholds for flooding frequency and floodplain aggradation are likely to vary among vegetation types and species;

- The natural variability in flooding frequencies due to El Niño Southern Oscillation (ENSO) added uncertainty to the effects threshold levels for physical stressors; and

- Modelling estimates of floodplain aggradation and flooding frequency over the large potentially affected area and prediction over a long time frame (decades) add uncertainty to risk screening estimates.

Aquatic Life

- Analyses of some analytes could not be conducted in either surface water or sediment due to a lack of exposure data (e.g., mercury, selenium);

- Analyses of other analytes were limited to a small portion of the study area due to limited concentration data (e.g., chromium, lead, nickel);

- Potential effects of particulate metals were not addressed due to a lack of effects data for most metals, but particulate copper is addressed in the DLRA;

- Water and sediment quality guidelines were not derived based on toxicity data for Papua New Guinea (PNG) species (although the guidelines tend to be conservative, it is unknown whether the guidelines are over- or under-protective of local species);

- Lack of sediment acid volatile sulphides:simultaneously extracted metals (AVS:SEM) data prevented evaluation of metal bioavailability in sediments;

- Potential risks posed by acid rock drainage were not evaluated.

Wildlife

- Exposure data were lacking for various stressors, locations and exposure pathways. Mercury in particular was not assessed in any way in the SLRA;

- Extrapolation of toxicity thresholds to species resident to the Ok Tedi/Fly River system from existing toxicity data in the literature introduced significant uncertainty;

- Bioconcentration and bioaccumulation factors were used to predict dietary exposure for some receptors (e.g., fruit bats) rather than measured chemical concentrations in tissues such as fruits and insects. This may be the most important uncertainty in this assessment; and

- For wildlife, it was assumed that 100 percent of measured metal concentrations were bioavailable in water, soil, sediment, and prey tissues.

CONCLUSIONS

Based on the information provided above, the release of mine-related materials to the Ok Tedi/Fly River system are responsible for physical and chemical stresses that pose current and future potential risks to the environment. Accordingly, further study in the form of a DLRA is warranted to more accurately quantify the risks and determine if proposed mine waste mitigation options effectively reduce these risks. Specifically, evaluation of both physical and chemical stressor effects on terrestrial vegetation, aquatic life, and wildlife in the Ok Tedi and middle Fly Rivers is needed. However, due to data and time constraints, copper was the only chemical stressor evaluated for aquatic life and no chemical stressors were evaluated for wildlife in the DLRA, despite a number of other stressors being identified for further evaluation in the SLRA.

Although some risks were predicted for human health, they are considered to be minor and do not warrant further investigation in the DLRA. However, additional monitoring of cadmium and lead levels in the Ok Tedi and Fly River estuary is recommended.

The SLRA provided focus to the DLRA by indicating which stressors of potential concern, assessment endpoints and receptors, and areas posed no significant risk and which ones require additional evaluation in the HERA. Specifically the SLRA:

- Found that direct risks to humans from mine-related chemicals are probably minor;
- Identified those chemical stressors that do not pose significant potential risk to ecological endpoints;
- Indicated that chemical stressors have the potential to pose risk to aquatic life and wildlife;
- Indicated that physical stressors have the potential to pose risks to aquatic life;
- Showed that physical stressors appear to be the predominant cause of vegetation dieback; and
- Indicated that the areas of potential adverse effects appear to be limited to the Ok Tedi and upper and middle Fly River, associated water bodies, and adjacent floodplain. The lower Fly River and estuary do not appear to be at risk.

1. INTRODUCTION

The purpose of this screening level risk assessment (SLRA) is to provide a screening of the stressors, human health and ecological endpoints, environmental pathways, and areas (river reaches) of concern to be retained for investigation in the Detailed Risk Assessment (DLRA). Descriptions of the screening process including assumptions, screening criteria, and data sources are provided.

The SLRA is comprised of four primary components: (1) problem formulation; (2) exposure characterisation; (3) effects characterisation; and (4) risk characterisation. Each of these components is discussed in the remainder of this report, followed by recommendations to reduce uncertainty in the risk assessment.

2. PROBLEM FORMULATION

Problem formulation is the planning phase of the risk assessment. It begins with defining the risk assessment objective. This is followed by describing human populations and terrestrial and aquatic ecosystems potentially at risk. An overview of the sources, characteristics, and fate of different potential stressors (e.g., chemicals) to human populations and ecological communities is then presented. The problem formulation concludes with a discussion of assessment and measurement endpoints followed by conceptual models showing how stressors and receptors interact in the environment.

2.1 RISK ASSESSMENT OBJECTIVE

Ok Tedi Mining Limited (OTML) and its consultants undertook a Human and Ecological Risk Assessment (HERA) of mine waste being released to the Ok Tedi/Fly River system. The purpose of the HERA was to determine how several proposed mine waste mitigation options affect potential risks to human health, and terrestrial and aquatic ecosystems. The HERA is part of an overall mine waste mitigation risk assessment (MWMRA) that also incorporates engineering and social risk analyses in selecting the optimal mine waste mitigation (MWM) option (Figure 1).

The HERA, and MWMRA as a whole, evaluated five specific MWM options:

Option A – Close mine FY2000.
Option B – Dredge to FY 2001 and store tailings to FY 2010.
Option C – Cease dredging operations in 1999.
Option DL – Dredge only to FY 2010, 15 million tonnes per annum (Mt/a).
Option DH – Dredge only to FY 2010, 19 Mt/a.

Option A involves cessation of mine operation and initiation of mine closure. Option B involves continued dredging of the dredge slot near Bige on the lower Ok Tedi to remove sand and reduce downstream flooding and aggradation to FY2001 and constructing a pipeline to transfer tailings from the mine to a location near Bige on the lower Ok Tedi. Option C involves cessation of dredging and continued mining operation. Options DL and DH involve continued dredging to the end of mine life and storage of dredged materials on the floodplain near Bige. A more detailed discussion of the mechanics and rationale for options can be found in OTML (1996). These five options were evaluated in a prospective manner in the HERA. Sediment transport and chemical fate models were used to project mine waste fate and transport in the system through the end of mine life and beyond. The potential risks to human health and the environment from mine waste were evaluated for the entire Ok Tedi/Fly River system including the estuary (Figure 2).

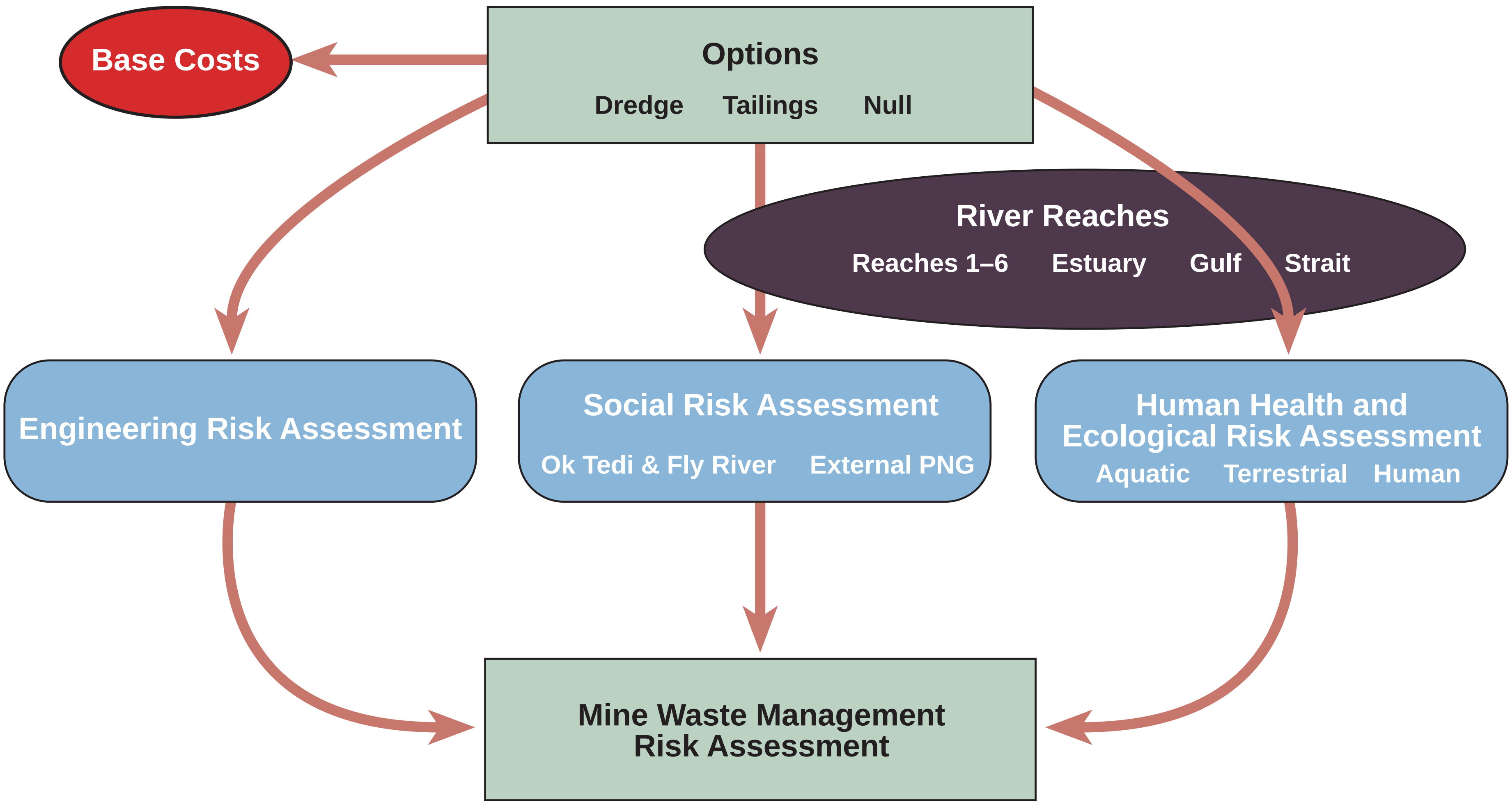


Figure 1.
Mine Waste Management Risk Assessment

Insert Figure 2 here.

Considering the large number of elements (i.e., different river sections, different components of mine waste potentially posing risk, and different receptors such as people, forest, aquatic life, and wildlife that may be at risk) in the HERA, it was impractical to evaluate all of these elements in significant detail. Rather, an SLRA was needed to focus the risk assessment on those geographic areas, receptors, exposure pathways, and mine waste constituents that may pose significant risk to human health and the environment. The SLRA used conservative assumptions regarding potential exposure and effects of mine waste constituents to effectively screen out those elements clearly not posing risk. This was the sole purpose of the SLRA. Because of the conservative assumptions and analytical manner in which an SLRA is performed, it is not intended to, nor capable of, quantifying the degree or extent of risk, or the comparative risks between MWM options. Comparisons between MWM options will be undertaken in the DLRA.

The advantage of performing an SLRA is that it provides a means to definitively demonstrate which elements should not be evaluated in detail during the risk assessment (Parkhurst et al. 1996; U.S. EPA 1998a). This, in turn, allows the risk assessors and managers to focus resources on quantifying to what degree, if any, risk is occurring for those elements that have not been screened out in the SLRA process. The remainder of this report describes the SLRA that used a standard, internationally accepted approach for assessing risks (CCME 1994; Environment Australia 1997; EC 1996; U.S. EPA 1998a).

2.2 HUMAN POPULATIONS, TERRESTRIAL, AND AQUATIC ECOSYSTEMS POTENTIALLY AT RISK

The following section provides an overview of the human populations, and terrestrial and aquatic ecosystems potentially at risk from mine-related stressors, as well as stressors not associated with the mine.

2.2.1 Human Populations

There are over 100 villages along the Ok Tedi/Fly Rivers from Kiunga downstream to Kiwai Island in the Fly River Estuary (Figure 2). The large geographic area considered in the risk assessment results in different uses and, hence, different levels of exposure to mine-related stressors across the study area. Despite the large geographic area, there are some generalities that apply to peoples of Papua New Guinea (PNG) that may be extrapolated to the people living in villages along the Ok Tedi/Fly River systems. There are some differences in stature, with highland peoples tending to be relatively shorter than coastal peoples (Frodin and Gressitt 1982). There also tends to be a corresponding relationship between height and body weight and it is expected that there are significant differences in body weight between lowland and highland peoples (CSIRO 1996). The physical stature of an exposed population has a direct bearing on exposure assessment. As described in Section 3.1.3, administered doses are calculated as a function of body weight. Based on the foregoing, it appears that the body weights of local villagers increase from the mountain regions towards the lowlands and estuaries. The precise reasons for these observations are not clear, but it likely has to do with increasing amounts of fish protein consumed in the lowlands and estuaries as compared to the highland areas.

Some common patterns in resource use are also anticipated. Traditional foods tend to be the same (e.g., root and tree crops), although techniques of cultivation may vary. The primary source of calories is usually sago (*Metroxylon* spp.) (Frodin and Gressitt 1982; Townsend 1974; Yok 1990a,b). Terrestrial species important in the diet include the pig (*Sus scrofa papuensis*) and cassowary (*Casuarius* spp.) (Frodin and Gressitt 1982; Bulmer 1968). Other species consumed include the cuscus (*Phalanger* spp.) and wallaby (*Macropus* spp.). The wild pig may tend to be more important in the lowland areas, while the cuscus occur at all elevations (Bulmer 1968). In addition, domesticated animals are often important, particularly the domestic pig. Finally, the raising of crocodiles (*Crocodylus* spp.) has been a reported food source in some villages and may occur in coastal or estuarine villages in PNG (Bolton 1976; Burgin 1980). These reports are consistent with what has been observed more recently in other areas of New Guinea (CSIRO 1996).

Dependence on riverine species as a food source is expected to be important for all villages along the Fly River. In coastal areas, dependence on marine aquatic species such as turtles (e.g., *Chelonia* spp.), dugong (*Dugong dugon*), sea urchins and crabs is expected to be important (Frodin and Gressitt 1982; Flew 1998). Additionally, fish species such as barramundi (*Lates calcarifer*) and large catfish are expected to be important sources of protein in the diet of villagers along the Fly River (Sorentino 1979).

Food crops are expected to be significant in the diet of people along the Ok Tedi and Fly River, as they are elsewhere in PNG (Bourke 1990; Morren 1977). Some of the commonly cultivated food crops throughout PNG are high starch staples such as sweet potato (*Ipomaea batatas*), taro (*Caladium colocasia*), yams (*Dioscorea* spp.), sago and banana (Blackwood 1940). The occurrence of these foods in gardens along the Fly River has been confirmed by Flew (1998).

2.2.1.1 Mountain Zone-Highland Villages

This river zone is generally defined as the reach between the location of mine waste discharge to the Ok Mani and the confluence of the Ok Tedi and Fly Rivers (D'Albertis Junction) some 180 km downriver. This river zone is less populated relative to the lower Fly River and estuary. Additionally, the Ok Tedi may not be a significant source of protein in this zone. This hypothesis is supported by the relatively high rates of malnourishment at villages near Konkonda (Flew 1998) that suggests a low-protein diet among Highland peoples. The primary food source in the Highland areas of PNG generally tends to be food gardens, with gardening in these areas generally being more intensive than in lowland areas (Yok 1989; Brown 1976).

Due to the lower population in the Highland areas near the Ok Tedi relative to the lower Fly River and estuary, and the expected lower intensity of river use, the potential for human exposure to mine-related stressors may be lower in this river zone relative to the lower Fly River and estuary.

2.2.1.2 Middle Fly River

On the middle Fly River, from D'Albertis Junction to Everill Junction, potential human exposures to mine-related stressors may occur from swimming or wading in the river and eating fish, reptiles or other aquatic life harvested from the river. The peoples along the middle Fly River and floodplain are considered a single potentially exposed population.

In addition to the main stem Fly River, the off-river water bodies (ORWBs) are a potential source of exposure in this zone. Thus, the same exposure routes (i.e., swimming, wading, and fishing) that apply in the main stem of the Ok Tedi/Fly River system also apply for ORWBs.

2.2.1.3 Lower Fly River and Estuary

Below Everill Junction, the land supports more forest and herds of rusa deer (*Cervus timoriensis*). A population of people lives along the Suki Lagoon, which is a relatively drier area than the middle Fly River. The same exposure pathways described for the middle Fly River are also applicable here. Several reports are available that provide some specific information to the lower Fly River, and data are available on the nearby Strickland River for comparative purposes. Flew (1998) states that protein foods such as wild game and fresh fish are abundant, and turtle and crocodile meat is sometimes consumed. Estimates of village-based fisheries and estimates of fish intakes in the range of 2-3 kg/person/week have been made (Swales 1997). Although fish intakes are estimated to be relatively high, the recent survey by Flew (1998) suggests that the predominant component in the diet along the Fly River and throughout the study area is sago, followed by bananas.

The Fly River estuary is the most heavily populated region of the Fly River. Food gardens are plentiful here, although sago remains the most important component of the diet (Flew 1998). Fishing is expected to be important in the diet here as it is in other coastal areas of PNG (Haines 1979). Similar to the middle and lower Fly River, exposures may occur by the same pathways described above. However, intake of aquatic foods (fish, invertebrates, and crocodiles) is expected to be greater here since they occur in greater abundance. Additionally, there may be some food items eaten here that do not occur further upriver (e.g., crabs, turtles).

2.2.2 Terrestrial Ecosystem

This section describes the flora and fauna of the terrestrial ecosystem. Because of the high mobility of many animal species and the dynamic interactions among components of the species occurring in different vegetation communities, the terrestrial ecosystem is defined broadly to encompass all terrestrial communities within the potentially affected portions of the Ok Tedi and Fly River basin.

2.2.2.1 Flora of the Terrestrial Ecosystem

The terrestrial ecosystem of the HERA study area is defined as the lands adjacent to the Ok Tedi/Fly River system that support emergent (e.g., mangrove, monsoonal savannah) or upland terrestrial (e.g., evergreen rain forest) vegetation and are potentially exposed to mine-related stressors. This area extends from more than 2,000 m elevation at the Ok Tedi mine to sea level in the Fly River estuary. In the upper reaches of the Ok Tedi basin, where the river is confined to a narrow channel, potential stressor effects are limited to a narrow band of vegetation bordering the channel. In the lower reaches of the Ok Tedi and along the middle reaches of the Fly River, potential effects extend onto the floodplain. Mine-related stressors may affect a broader band of terrestrial vegetation along the middle and lower Fly River, where the floodplain is wide.

Several vegetation types occur in the study area. These range from lower montane rain forest in the vicinity of the mine through lowland rain forest and open savannah in the middle reaches of the Ok Tedi/Fly River systems, and finally to mangroves at the mouth of the Fly River. Topographic changes and their effect on rainfall gradients, cloud cover, and temperature are largely responsible for the differences in vegetation. The variety of vegetation in turn provides habitat for diverse assemblages of animal species, many of which are endemic to New Guinea.

Lower montane forest occurs above 1,000 m in the upper reaches of the Ok Tedi basin in the vicinity of the Ok Tedi Mine. The forest is mixed evergreen with a canopy height of 25 to 30 m (Gregory 1995) and, in contrast to forests at lower elevations, there is a significant accumulation of leaf litter and logs. Mosses and ferns are abundant and cover much of the ground and interior forest vegetation. Tree ferns and epiphytes are common. The lower montane forest supports a variety of bird, reptile, and amphibian species and the most diverse mammal assemblage in New Guinea (Flannery 1995).

Foothill forest forms a distinct vegetation type on steep slopes and shallow, unstable soils below 1,000 m elevation. The canopy is generally less than 30 m high, but in places emergent *Araucaria* trees can attain heights of 50 m. Cloud cover may be a more important factor than temperature in influencing vegetation change with elevation. A few mammal species are entirely restricted to this zone, but other species also occur in the adjacent montane zone that extends into the lowland alluvial floodplain forest (Flannery 1995).

Lowland alluvial floodplain forest covers the broad plains below the foothills along the lower Ok Tedi and middle reaches of the Fly River. The forest is tall (canopy height 30-35 m), floristically rich, and probably primarily of Asian origin (Flannery 1995). Palms are common and epiphytes are plentiful, but leaf litter is sparse. These forests support only a limited endemic mammal fauna. Only the grey dorcopisis (*Dorcopsis luctuosa*), a few murid rats, and some bats are found in this habitat type (Flannery 1995).

Much of the vegetation along the lower Ok Tedi to D'Albertis Junction is characterised by abandoned garden sites, secondary regrowth forest, and mature lowland rain forest. Most of the species of the rain forest and secondary regrowth are tolerant to flooding, probably up to 15 percent of the time (OTML 1996). South of D'Albertis Junction, the floodplain widens on both sides of the Fly River and forest habitat gives way to a mosaic of open canopy forest and open grass-covered swamps with reed (*Phragmites karka*), sedges (e.g., *Schoenus* spp., *Scleria* spp.), pandanus (*Pandanus* spp.), wild pitpit (*Saccharum robustum*) and other more flood tolerant plants dominating.

The mixed savannah of monsoonal southwest PNG is a seasonal swamp characterised by a variety of grasses. Common grass species include: kunai grass (*Imperata cylindrica*), *Ophiuros tongcalingii,* and *Ischaemum barbatum*. Fires are frequent during the dry season, but much of the area is flooded during the wet season. On islands, and at the edge of the flood plain, paperbark tree (*Melaleuca* spp.) and other flood tolerant species occur. Much of the savannah habitat in Papua

New Guinea is in a state of flux because of changing water tables and man-made fires (Gillison 1983).

Extensive mangrove areas occur along the lower Fly River below Everill Junction and throughout the Fly River estuary. As in other parts of the world, these areas provide important nesting habitat for birds and nursery areas for a broad range of fish and invertebrate species. With roots in the water and stems and leaves in the terrestrial environment, they play a keystone role in both the aquatic and terrestrial ecosystems of the estuary and tidally influenced portions of the Fly River. Mangroves also influence currents and sedimentation processes as well as provide a significant source of organic carbon (i.e., nutrients) to the ecosystem.

2.2.2.2 Fauna of the Terrestrial Ecosystem

Unlike the animal communities in typical rain forests in other parts of the world, there are no cats or monkeys in New Guinea. The largest species in the ecosystem are snakes (pythons) and birds (cassowaries); the latter can attain body weights of 60 kg (Beehler et al. 1986). The diversity of vertebrate species of New Guinea is considered impoverished compared to those of larger continental areas (Bourliere 1983a). Insect diversity is high, and many of the species in New Guinea rain forests occur in the forest canopy (Sutton 1989).

Mammals (with the exception of the dugong) can be considered exclusively terrestrial in terms of habitat usage. However, extensive flood plains provide a broad interface between the terrestrial and aquatic ecosystems of the Ok Tedi/Fly River system, making it difficult to delineate precise boundaries between aquatic and terrestrial ecosystems for many birds and reptiles. Numerous species are semi-aquatic and/or feed across these ecosystem boundaries. Kingfishers (Alcedinidae), storks (Ciconiidae), herons and egrets (Ardeidae), Australian pelican (Pelecanidae), and cormorants (Phalacrocoracidae) feed primarily on fish and invertebrates in the aquatic ecosystem. Whimbrel (*Numenius phaeopus*), sandpipers (*Calidris* spp.), and other shorebirds obtain their food from shallow sediments in marshes, mudflats, and along beaches. Crocodiles, turtles, and monitor lizards are semi-aquatic; they occur throughout the floodplain and at the edges of waterbodies.

As discussed in Section 2.5.4.3, risks to all fauna except fish and aquatic macroinvertebrates are assessed in a similar manner. Accordingly, for this reason and for simple practicality, these fauna (collectively called wildlife) are grouped together as part of the terrestrial ecosystem regardless of the habitat (terrestrial or aquatic) they may actually occupy.

<u>Mammals</u>

The mammalian fauna of New Guinea are distinctive, but show close affinity to Australian fauna. Marsupials such as bandicoots (*Microperoryctes* spp., *Peroryctes* spp.), wallabies and tree kangaroos (*Macropus* spp.), and cuscus are typical terrestrial mammals on New Guinea and occur in the study area. Echidnas are also present, though generally uncommon. Bats are diverse and include species that feed on insects, fruit, and nectar (Flannery 1995; OTML 1996). Savannah habitat along the lower Fly River supports a large and diverse assemblage of mammals. The delicate mouse (*Pseudomys delicatulus*) and brush-tailed rabbit rat (*Conilurus penicillatus*) are restricted to this habitat. Both are reported from the trans-Fly River plains (Flannery 1995).

Pigs and two species of deer introduced to New Guinea now provide a major source of animal protein for the human population throughout the island. Flannery (1995) reports that the wild dog (*Canis familiaris*), which was introduced to New Guinea about 2,000 years ago, and humans are the only large mammalian predators among a diverse faunal assemblage of herbivores, omnivores, and insectivores that comprise over 200 species of mammals.

Birds

New Guinea has a total of about 650 bird species of which 568 are breeding land and freshwater species (Rand and Gilliard 1968). Distinctive groups include bowerbirds, cassowaries, and more than three dozen species of birds of paradise. Raptors (e.g., eagles, hawks, and owls) range throughout the study area. Shorebirds and wading birds are abundant throughout the floodplain of the middle and lower Fly River. Species in each of these groups occur in potentially affected habitats in the study area (Beehler et al. 1986; Gregory 1995). The forests of New Guinea, including much of the study area, are also rich in species of fruit pigeons that are important seed dispersers in tropical forests (Janzen 1983).

Reptiles and Amphibians

Amphibians and reptiles are abundant in habitats throughout the study area. Tree frogs (Hylidae) dominate the fauna and are represented by numerous species. True frogs (Ranidae) are also present, as are old world tropical tree frogs of the family Microhylidae (Darlington 1957; Zweifel 1972). There are no salamanders or caecilians on New Guinea (Darlington 1957). Lizards and snakes occur in all terrestrial habitats found in the study area. Pythons, boas, and poisonous snakes, such as the death adder, occur in both forest and savannah habitats (Mehrtens 1987). Agamid lizards and goannas, a type of monitor lizard, are widely distributed in the study area.

Invertebrates

Invertebrates play a central role in the structure and function of terrestrial ecosystems of the study area. The group includes predators (e.g., spiders), folivores (e.g., Lepidoptera larvae), grazers (e.g., snails), pollinators (e.g., butterflies, bees), and detritivores (e.g., mites). They occupy all strata in the forest and play significant roles in the food web. While many species inhabiting tropical rain forests remain undescribed, insects represent the highest diversity of any animal group.

2.2.3 Aquatic Ecosystem

This section describes the aquatic biology of the Ok Tedi/Fly River system. The aquatic habitats common to the system, as well as the different types of organisms common to these habitats are discussed.

2.2.3.1 Aquatic Habitats

As described in Section 2.4.1, drainage from the valleys surrounding the south side of the mine flows into the Ok Mani, which discharges into the Ok Tedi. In turn, the Ok Tedi joins with, and becomes, the Fly River until it drains into the Gulf of Papua. Given the elevation change from the mine to the coast, the river system supports a wide range of habitats from the Ok Mani to the Gulf of Papua.

The Ok Mani and Ok Tedi are typified by steep gradients over limestone and mudstone substrata, and flow through rainforest riparian vegetation. In the lowland reaches, the Fly River meanders through extensive floodplains with either rainforest vegetation (upper parts) or flooded swamps and grasslands (lower middle Fly River). The Fly River is tidally influenced at its confluence with the Strickland River, almost 400 km upstream from its mouth. These lower reaches of the Fly River become more deeply incised than upstream, and the floodplain is reduced. This reach is also characterised by rainforest vegetation, with a mix of open woodlands. The lower reach is also influenced by strong tidal bores, resulting in a scoured bank profile.

2.2.3.2 Aquatic Organisms

Aquatic habitat varies distinctly from the upland reaches to the estuary. Accordingly, the types and relative abundance of organisms also are variable. The aquatic organisms common to the various river reaches are described separately below. The aquatic biology of the Ok Tedi/Fly River system has been summarised in detail by R&D Environmental (1998).

Aquatic Plants/Algae

There do not appear to be many data on the aquatic vegetation of the Ok Tedi/Fly River system. Power et al. (unpublished data), as cited in Storey (1998), sampled macrophytes and algae from the floodplain at Lake Pangua. Organisms collected included *Utricularia* spp. (bladderwort), *Azolla* spp. (fern), periphyton and cyanobacteria. Coarse grasses also were collected, but were not considered an important carbon source for fish. Stauber and Apte (1996) have summarised phytoplankton in the Fly River and Strickland River systems. They identified 12 genera of cyanobacteria, 16 genera of diatoms, 40 genera of green algae, and multiple genera of other phytoplankton, including euglenophytes and dinophytes.

Macroinvertebrates

Macroinvertebrate sampling has been limited in most habitats, but some data are available. In the upland streams, mayflies from the families Baetidae and Leptophlebiidae, caddisflies (Hydropsychidae, Leptoceridae), flies (Simuliidae, Chironomidae, Tabanidae), and prawns (mostly *Macrobrachium handschini* and *M. rosenbergii*) dominate the invertebrate communities. Based on the limited invertebrate data for the middle Fly River, the fauna in this reach appear to be numerically dominated by Diptera, Ephemeroptera, and Crustacea. The greatest biomass in the middle Fly River was contributed by large burrowing mayflies (*Plethogenesia papuana*) and prawns (*M. equidens*, *M. handschini*, *M. lorentzi*, *M. mammilodactylus*, *M. papuanum*, *M.*

rosenbergii, *M. novaehollandiae*, and *M. weberi*). In the lowland river channels, macroinvertebrate assemblages are generally limited except where burrowing mayflies are locally abundant. The macroinvertebrate assemblages on the floodplains are diverse, but dominated by mobile, facultative species that can rapidly colonise these ephemeral habitats. The Fly River estuary is suspected to be an important nursery ground for mud crabs (*Scylla serrata*) and banana prawns (*Penaues merguiebsis*) (OTML 1997a).

Fish

The fish fauna of the Ok Tedi/Fly River system are extremely diverse but not well understood except for barramundi (the only commercially important species in the river system). The following discusses the fish species common to the Ok Tedi, middle Fly River, lower Fly River/estuary, and the Gulf of Papua.

In the upper Ok Tedi, insectivorous and frugivorous fish appear to be most dominant. These include the green catfish (*Arius latirostris*), the river garfish (*Zenarchopterus novaeguineae*), and the common rainbowfish (*Melanotaenia splendida*) (OTML 1983; 1984a,b; 1985a,b; 1986). The fish species in the lower Ok Tedi are more diverse than in the steeper reaches of the upper Ok Tedi. The most numerous fish species in the lower Ok Tedi are herrings (*Nematalosa* spp.), catfish (including *Cochlefelis spatula*, *Arius macrorhynchus*, *A. taylori*, *A. berneyi*, and *A. latirostris*), and carnivores such as the narrow-fronted tandan (*Neosilurus ater*), barramundi (*Lates calcarifer*), and the giant perchlet (*Parambassis gulliveri*). The fish that contribute the greatest biomass include *L. calcarifer*, *C. spatula*, *A. leptaspis*, *A. latirostris*, *A. taylori*, *A. macrorhynchus*, oxeye herring (*Megalops cyprinoides*), *A. berneyi*, narrow-fronted tandan, and Papuan bass (*Lutjanus goldei*) (R&D Environmental 1998).

Sixty fish species from 24 families have been collected from the upper middle Fly River. The species are similar to those found in the lower Ok Tedi, but also includes sawfish (*Pristis microdon*), greenback mullet (*Liza subviridis*), and climbing perch (*Anabas testudineus*) (R&D Environmental 1998). Gill net catches in the upper middle Fly River are numerically dominated by *Nematalosa* spp., *Arius berneyi*, barramundi, *Arius leptaspis*, *N. ater*, and *Arius macrorhynchus* (OTML 1983; 1984a,b; 1985a,b; 1986). Seine catches are dominated by the sailfin perchlet (*Ambassis agrammus*), common rainbowfish, *P. gulliveri*, the toothed river herring (*Clupeoides papuensis*), and *Nematalosa* spp. (OTML 1983; 1984a,b; 1985a,b; 1986). In the middle Fly River, a piscivorous food chain appears to dominate as the invertebrate assemblages decrease in abundance. For example, the biomass of the lower middle Fly is dominated by barramundi and a catfish (*A. leptaspis*), both fish-eaters, and herring (prey fish). Since herring also are abundant in upper and lower middle Fly River floodplains, it is likely that they feed in the ORWBs and may only use the main river channel for travelling from one ORWB to another. The bottom of the food chain for organisms in this reach appears to be macroinvertebrates and detritus in the ORWBs on the floodplain surrounding the river. Barramundi are particularly plentiful in this reach. Of all fish caught in gill net sampling from 1983 to 1996, barramundi biomass was three times greater than the next greatest contributor (R&D Environmental 1998).

Most fish samples from the floodplain adjacent to the middle Fly River have been collected in oxbow lakes due to the difficulty in sampling from the flooded forest. In two oxbow lakes in this reach, catches were dominated by herring. Other common species include *A. berneyi*, longtoms (*Strongylura kreffti*), anchovy (*Thryssa rastrosa*), *A. leptaspis*, *Toxotes chatareus*, *Scleropages jardini*, *Megalops cyprinoides*, *Amniataba percoides*, narrow-fronted tandan, and barramundi (R&D Environmental 1998).

The fish assemblages in the lower Fly River are more diverse than in any other reach (75 species from 32 families). These species include predominantly estuarine species, such as bull shark (*Carcharhinus leucas*), scat (*Scatophagus argus*), toadfish (*Chelonodon patoca*, *Marilyna meraukensis*), and spaghetti eel (*Moringua penni*). Freshwater specialist species have also been collected in this reach, including Weber's mudskipper (*Periopthalmus weberi*), spoon-snouted catfish (*Doiichthys novaeguineae*), Bintuni goby (*Stenogobius lachneri*), and brown gudgeon (*Eleotris fusca*). Gill net samples in the lower Fly River are numerically dominated by barramundi, *A. leptaspis*, *A. carinatus*, *Thryssa scratchleyi*, and *C. danielsi*, and gravimetrically dominated by barramundi, *A. augustus*, and *A. leptaspis*. It should be noted that most sampling has been conducted at the upstream end of the reach, so these results are biased in that direction (R&D Environmental 1998).

2.3 STRESSORS OF POTENTIAL CONCERN

A stressor can most generally be defined as anything that negatively affects human health and/or the environment. Stressors can be physical (e.g., fire, flooding), chemical (e.g., metals, pesticides), or biological (e.g., disease, limited food resources). The objective of this assessment is to evaluate risks associated with mine waste discharge. All three types of stressors may be directly or indirectly associated with this activity.

Stressors of Potential Concern (SOPCs) are those stressors that are directly or indirectly derived from mine waste discharge. In this risk assessment, the primary source of stressors is waste being released to the Ok Tedi/Fly River system from the mine located on top of Mt. Fubilan. It is important to distinguish that SOPCs do not necessarily pose significant risk to human health or the environment; they only have the potential to pose risk. The overall purpose of the SLRA is to rule out SOPCs not posing risk, so more detailed attention can be paid to the SOPCs that may be posing significant risk.

2.3.1 Mine-Related SOPCs

OTML has been operating an open pit mine on top of Mt. Fubilan in the Western Province of Papua New Guinea since 1984 (Lee et al. 1997). The ore body consisted of copper-gold mineralisation associated with porphyry (granitic) intrusions and was structured such that a gold bearing cap was overlying the copper-bearing material. OTML operated a standard cyanide leaching circuit to process the gold bearing material until 1989 after which time the copper has been processed using a flotation mill. The mill also is located on top of Mt. Fubilan immediately adjacent to the pit.

Throughout the mine life, mine waste in the form of overburden material (i.e., waste rock), and tailings from the mill have been released to the Ok Tedi/Fly River system. Historically, mine waste has been dumped from the north and south sides of the mountain but has only been dumped from the south side since 1989. Mine waste released from the south side sloughs down the flank of Mt. Fubilan and into the Ok Mani drainage that joins the Ok Tedi approximately 5 kilometres downstream (Figure 3).

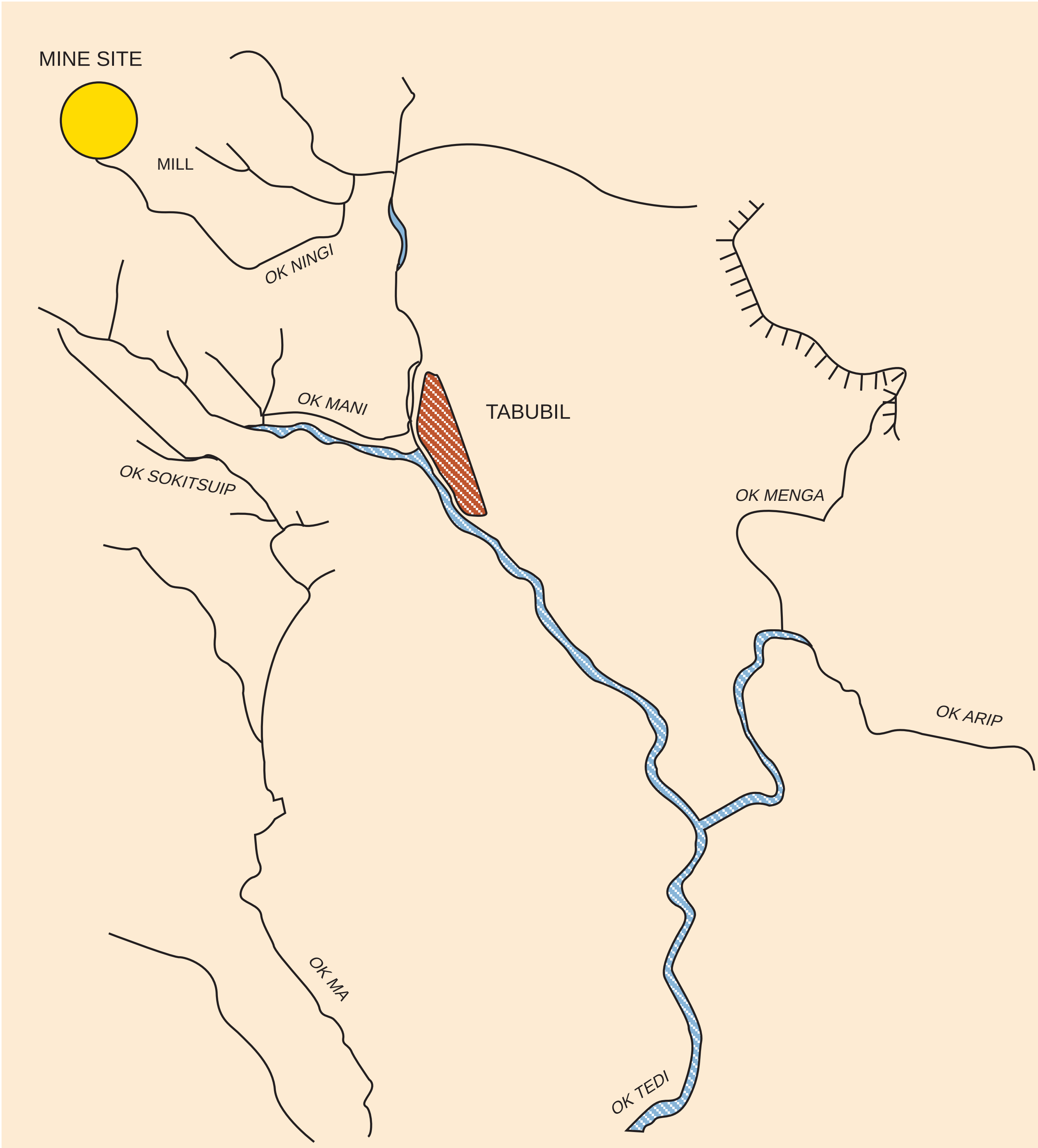
2.3.1.1 Physical Stressors

The most significant component of the mine waste is the solids associated with waste rock and tailings. Approximately 44 and 30 million tonnes of waste rock (not including Harvey Wall erosion) and tailings, respectively, are released to the Ok Mani on an annual basis (Lee et al. 1997) as of 1997. The amount of waste rock is expected to increase to 55 Mt/a by 2000 followed by a gradual decline from 2003 to end of mine life. Tailings production will remain at approximately 30 Mt/a throughout mine life. This material is transported down the river system with finer materials being transported the furthest downstream. This process is described in greater detail in Sections 2.4.1 and 2.4.2.

The primary physical stressor on the terrestrial ecosystem from mine waste discharge is riverbed aggradation (sedimentation) of the channels carrying mine waste. Tailings and overburden rock are deposited as sediment in channels and on floodplains adjacent to the river channels below the mine. In locations where the additional accumulation of mine waste deposited as sediment becomes sufficiently deep on the floodplain, it buries the organic litter and ground layer vegetation, smothers tree roots, and drowns soil flora and fauna. The increased volume of mine waste produces secondary stress in the form of increased flooding on the floodplain and on areas adjacent to sediment deposition. It also produces some scouring along main channels and across the necks of meanders.

Similar to the terrestrial ecosystem, riverbed aggradation is a significant stressor to the aquatic ecosystem. Aggradation buries fish and aquatic macroinvertebrate habitat. Throughout the river, backwaters, snags, and riffles (in upper reaches) are buried, eliminating critical habitat components. Even where aggradation is not sufficient to bury habitat, addition of mine-derived materials low in nutrients (e.g., organic carbon) will dilute nutrient concentrations in the system and potentially lead to deleterious effects. In addition to aggradation, the increased solids load to the system substantially increases total suspended solids (TSS) in the water column. This increase has the potential to cause deleterious effects on aquatic life. These effects are likely to be less pronounced in the Ok Tedi/Fly River system compared to many of the world’s river systems, because it naturally has a relatively high TSS load, to which aquatic organisms are adapted.

Lastly, for aquatic life, oxygen deprivation (hypoxia) may also stress or kill organisms. Although dissolved oxygen (DO) levels regularly decline to extremely low levels in the middle Fly River, particularly on the floodplain, the mine may be exacerbating this stress.



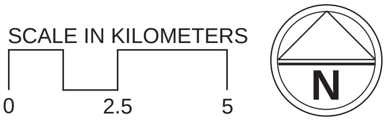


Figure 3.
Ok Mani and Upper Ok Tedi Drainage

2.3.1.2 Chemical Stressors

Chemical stressors on human populations and the terrestrial and aquatic ecosystems include the metals and mine-related chemicals present in the tailings and other mine wastes that have been discharged into the surrounding environment. The list of potential chemical stressors for each of these receptor groups is summarised in Table 1. Their environmental fate and transport are discussed in Section 2.4.3.

Table 1. Chemical stressors of potential concern.

Aquatic	Terrestrial	Human Health
Aluminium	Aluminium	Aluminium
Arsenic	Arsenic	Arsenic
Cadmium	Cadmium	Cadmium
Chromium	Chromium	Chromium
Copper	Copper	Copper
Iron	Iron	Iron
Lead	Lead	Lead
Manganese	Manganese	Manganese
Mercury	Mercury	Mercury
Molybdenum	Molybdenum	Nickel
Nickel	Nickel	Selenium
pH	Selenium	Silver
Selenium	Silver	Zinc
Silver	Zinc	
Zinc		
Mill Reagents		

2.3.1.3 Biological Stressors

Vegetation dieback along the channel and in the floodplain is a secondary stressor on human populations, terrestrial animals, and aquatic life that use the vegetation as either habit and/or food. The absence of ground vegetation, canopy cover, and fruit reduce the availability of food in affected areas. Where these areas are sufficiently large, they can adversely affect the environment.

2.3.2 Other Stressors

In addition to mine-related stressors, other stressors not related to the mine are significant in both number and their potential impact on human health and the environment. For human populations, disease, malnourishment, and poor hygiene probably represent the greatest potential stresses. For the environment, natural weather phenomenon such as El Niño, natural and man-made fire, introduced species, and increasing human use of the environment represent stresses that may be as great, or greater than those posed by the mine.

All of these natural and anthropogenic disturbances provide a dynamic background against which to evaluate mining effects on the system. It is problematic to assess risks from one source without considering risks from other sources. Overlapping stresses can lead to unexpected changes (Bright 1999). However, because the specific objective of this risk assessment is to evaluate the relative risks associated with the different proposed mine waste mitigation options, only mine-related stressors are considered in this study. Additionally, the assessment of cumulative mine-related risks is beyond the scope of the SLRA but is addressed in the DLRA.

2.4 FATE AND TRANSPORT OF PHYSICAL AND CHEMICAL STRESSORS

2.4.1 General Overview of Ok Tedi/Fly River System Hydrology

The Fly River catchment covers approximately 76,000 km^2 and is drained by two major rivers – the Fly and the Strickland (Figure 2). The headwaters of the catchment are between 2000 and 3500 m elevation and occur along the northwest trending cordillera of New Guinea. Heat and moisture associated with the inter-tropical convergence zone encounter orographic lifting along the cordillera providing rainfalls in excess of 10 m per annum near Mt. Fubilan, decreasing to 8 m at Tabubil, 4.5 m at Kiunga and 2 m along the coast. These high rainfall amounts in the upper catchment support the substantial flows observed in the Fly and Strickland Rivers and the large volume of water found across the Fly River floodplain.

The Ok Tedi headwaters begin near the Hindenburg Wall, a 1000 m limestone escarpment. The river flows past the mine and then drops steeply out of the mountains for approximately 160 km until it reaches the town of Ningerum (80 m elevation and average flow of 270 m^3/sec). At this point, the river gradient shallows dramatically. After a further 80 km, the Ok Tedi joins the Fly River at D'Albertis Junction (30 meters elevation and 2,200 m^3/sec average flow below the confluence) and then meanders another 400 km before being joined by the Strickland at Everill Junction (5,400 m^3/sec average flow below the confluence). The river then travels an additional 400 km before entering the Gulf of Papua.

The middle Fly River (between D'Albertis and Everill Junction) has an extensive floodplain covering approximately 5,500 km^2. The floodplain consists of a myriad of tie channels, oxbow lakes, blocked valley lakes, swamps, lagoons and periodically flooded forest. A total of 29 major ORWBs are distributed largely across the lower half of the middle Fly River. This floodplain stores large volumes of water that flow in and out of the river on a seasonal basis. Below Everill Junction, the Fly River becomes increasingly tidally influenced. Hypoxic conditions in ORWBs occur regularly and naturally (Yaru 1998). The biological productivity and resulting input of organic matter may determine the magnitude of hypoxia (Yaru 1998). The mine may also affect DO levels as increased TSS may inhibit photosynthesis in aquatic plants and algae. In addition, increased aggradation may trap waters on the floodplain for extended periods, thereby increasing the residency time of water with high organic matter.

2.4.2 General Overview of Ok Tedi/Fly River System Sediment Transport

The high rainfall and steep gradients in the upper catchment naturally provide for significant sediment transport in the Ok Tedi/Fly River system. Approximately 5 Mt/a of sediment are eroded into the Ok Tedi via these natural processes. In addition to the continual erosional processes, periodic massive landslides characterise the system. Landslides on the scale of the 1989 Vancouver Ridge Landslide (60 million cubic meters) have an annual recurrence probability of 3% in the upper Ok Tedi catchment (MUDS 1991).

The mine contributes an additional 74 Mt/a of solids to the river system via the waste rock and tailings dump at Mt. Fubilan. This material is released to the Ok Mani, a headwater tributary of the Ok Tedi (Figure 3). Additional material associated with valley wall erosion as the mine waste travels down creeks on the flanks of Mt. Fubilan and into the Ok Mani further increases the sediment load. Approximately 40 percent of the mine waste and valley wall erosion material has been retained in the Ok Mani drainage with the remaining 60 percent being released to the Ok Tedi (OTML 1996).

The increased sediment loads to the Ok Tedi/Fly River system have resulted in two physical effects – 1) substantial aggradation of the riverbed and subsequent flooding of adjacent areas, and 2) a significant increase in TSS concentrations in the system.

Aggradation of mine waste is the ultimate physical stressor to the terrestrial ecosystem resulting from mine waste discharge. Aggradation of this material in the stream channel (riverbed aggradation) causes water levels to rise, flooding adjacent terrestrial habitat and prolonging flooding in seasonally flooded areas adjacent to water channels. In river reaches where the floodplain is confined between terraces and the main channel meanders, increased water levels can lead to bank scouring and new channels where the river cuts across the narrow necks of meanders. This scouring increases the solid materials in the river through the addition of eroded soil and organic debris (e.g., trees). Scouring effects of this type have been observed in the lower Ok Tedi River and the upper portion of the middle Fly River.

In the upper reaches of the Ok Tedi basin, where the river is confined to a narrow channel, mine waste accumulates to a depth of several meters. Riverbed aggradation raises the water level, causing increased flooding in a narrow band adjacent to the stream channel. The resulting effects on the terrestrial ecosystem are limited to this narrow band. Prolonged flooding in the main channel can cause increased and prolonged flooding at the mouths of tributary streams, causing vegetation dieback in areas not directly exposed to mine waste sedimentation. In the lower reaches of the Ok Tedi, and along the upper middle reaches of the Fly River, increased bed aggradation causes flooding to extend onto broader areas of the floodplain.

Aggradation of mine waste also results in increased sediment deposition along the channels of the Ok Tedi/Fly River system. In the upper reaches of the Ok Mani and Ok Tedi, material is relatively unsorted and ranges in size from boulders to silt. With decreasing gradient and in areas where the channel is less confined, generally smaller particles are carried downstream. However, during major flood events, larger particles can be carried considerable distances in the stream channel.

In river reaches where the river periodically spreads onto the floodplain, smaller sediment particles are deposited on the floodplain. This floodplain aggradation buries natural vegetation, gardens, and surface litter. These materials contain elevated concentrations of metals and other chemicals associated with mine waste and are low in nutrients.

Aggradation also results in significant loss of aquatic habitat in the mainstem river, particularly in the Ok Tedi. Further downstream in the middle Fly River, some filling of ORWBs has been observed due to the increased sediment load although actual blocking of the channels does not appear to be taking place. In addition to the habitat loss, the increased TSS concentrations have the potential to impact the aquatic ecosystem. Elevated TSS may be directly affecting aquatic life via gill abrasion and clogging. Additionally, the decreased light penetration into the water column resulting from elevated TSS may be reducing primary productivity of phytoplankton.

2.4.3 Factors Affecting Fate, Transport, and Bioavailability of Chemical Stressors

2.4.3.1 Water

The following discusses the general fate of mine waste chemical constituents in surface water. Because some of the chemicals share common fate processes, they are divided into three general classes: 1) divalent metals (e.g., cadmium, copper, lead); 2) metals/metalloids with important and naturally occurring organic forms (i.e., arsenic, mercury, selenium); and 3) other chemical stressors (e.g., aluminium, antimony, pH, mill reagents).

Divalent Metals

The divalent metals evaluated in the SLRA include cadmium, copper, lead, nickel, and zinc. In the water column, divalent metals may dissolve, precipitate as insoluble compounds, adsorb to suspended particles, or form complexes with organic carbon. Precipitated metals and metal complexes adsorbed to suspended solids may eventually settle onto the sediment. The amount of dissolved metal that precipitates depends on, among other factors, pH, redox conditions, and the ions present in the water (Bodek 1988; Eisler 1985, 1993, 1997). Callahan et al. (1979) stated that precipitation might be important in removing dissolved metals from the water column in severely contaminated areas, but adsorption to suspended particles is generally a more important removal mechanism. Divalent metals tend to adsorb to mineral surfaces, hydrous metal oxides, and organic materials (Callahan et al. 1979).

The bioavailability of divalent metals in the water column to aquatic organisms depends on their form. In general, dissolved divalent metals are considered to be the most bioavailable to aquatic organisms (Spry et al. 1988; Evans et al. 1988; Thomas et al. 1983; Part and Svanberg 1981; Hodson et al. 1978; Davies et al. 1976; Davies 1976). In freshwaters, water hardness also influences the bioavailability of divalent metals (i.e., the higher the hardness, the less bioavailable the metal). Cations such as calcium and magnesium, which constitute “hardness”, compete with divalent metals at organism binding sites (e.g., the gill). In the case of copper, alkalinity has a greater influence on bioavailability than hardness (Nelson et al. 1986; Andrew 1976; Shaw and Brown 1974). The effect of alkalinity on copper bioavailability is different than the effect of hardness in that it is a function of carbonate ions binding with copper in solution and forming less bioavailable complexes.

Finally, it has been demonstrated that all of the divalent metals are capable of forming complexes with dissolved organic matter (DOM) that have limited bioavailability (Meador 1991; Playle and Dixon 1993; Winner and Gauss 1986). Studies by CSIRO, on behalf of OTML, have demonstrated that the bioavailability of dissolved copper in the Fly River system is substantially reduced by complexation with DOM (CSIRO 1997). For example, algae bioassays using site water samples were non-toxic at dissolved concentrations that should have been toxic if the copper were in a bioavailable form. This indicates that most of the dissolved copper was present in the form of copper-organic complexes. Similar but weaker complexes likely exist for other divalent metals in the Fly River system, but this has not been demonstrated to date.

Arsenic, Mercury, and Selenium

Arsenic, mercury, and selenium were grouped separately because organic forms of these metals/metalloids are important in the discussion of their environmental fate. Some of the divalent metals discussed above also can exist in organic forms, but these are not considered as important in the natural environment. Common organic forms of arsenic include methanearsinic acid and dimethylarsinic acid, while the most common organomercurial is methyl mercury. Inorganic selenium can be transformed into seleno-amino acids and methyl selenium compounds.

In the water column, arsenic occurs in both inorganic and organic forms, although inorganic arsenic (V) is the most common species (Eisler 1988). Arsenic (V) is favoured under conditions of high DO, basic pH, high Eh, and reduced content of organic material; the reverse conditions favour the formation of arsenites and arsenic sulphides (Eisler 1988). As for the divalent metals discussed above, the fate of arsenic in surface waters is dependent on the chemical and physical characteristics of the water. For example, arsenates are adsorbed by anion retention sites (e.g., amino groups) on colloidal humic material under conditions of high organic carbon content, low pH, low phosphate, and low mineral content (Eisler 1988). Also, arsenates may coprecipitate with, or absorb on, hydrous iron oxides or from insoluble complexes with calcium, sulphur, aluminium, and barium (Eisler 1988).

The majority of mercury present in water usually is in an inorganic form. For example, only 3 percent of the mercury in water samples collected in an estuarine bay was found to be methylated (Alcoa 1996). The fate of mercury in water is governed by many processes, including complexation/dissociation, precipitation/dissolution, and adsorption/desorption (Mason et al. 1995a; Fitzgerald et al. 1994; Mason et al. 1993). These processes are in turn governed by pH, Eh, salinity, the availability of ligands, dissolved and particulate iron and manganese concentrations, and the size and composition of suspended solids (Fitzgerald et al. 1994; Mason et al. 1993; Sadiq 1992; Bodek 1988; Callahan et al. 1979). In natural waters, it is generally accepted that the bioavailability of mercury to aquatic life is best predicted on the basis of the free cationic form of mercury (Hg [II]) and certain neutral, lipophilic mercury complexes (e.g., $HgCl_2^o$) (Mason et al. 1995b; Sadiq 1992; Campbell and Lewis 1988; Gutknecht 1981; Nriagu 1979). In addition, the bioavailability of methylated mercury is also high (D'Itri 1990; Sadiq 1992; Mason et al. 1995b). Neutral complexes (e.g., $HgCl_2$ and CH_3HgCl) are the most bioavailable complexes (Gutknecht 1981; Mason et al. 1995b).

The majority of the selenium in aerobic waters is present as either selenium (IV) or selenium (VI) (both are inorganic forms). Selenite adsorbs to manganese dioxide, while both selenite and selenate adsorb to iron oxyhydroxide (Balistrieri and Chao 1990). In addition, both selenite and selenate adsorb to clays such as kaolinite and montmorillonite (Bar-Yosef and Meek 1987). Organic selenides may also be present in surface water as a result of metabolism by aquatic organisms, but these would be present at much lower concentrations than inorganic forms.

Other Chemical Stressors

This grouping consists of metals and metalloids such as aluminium, antimony, chromium, iron, molybdenum, and silver. The fate of each of these metals/metalloids in surface water is dependent on their chemical form, which in turn is dependent on the chemical and physical characteristics of the water. As for the other metals/metalloids, the amount and types of solids in the water, as well as water quality parameters such as pH and Eh will influence their partitioning onto solids. Precipitation will also play an important role in determining the fate of some of these metals, such as chromium (U.S. EPA 1985d). The dissolved fraction is typically the bioavailable fraction for metals/metalloids. However, for aluminium, there is conflicting evidence that dissolved aluminium is the bioavailable form (U.S. EPA 1988a).

Two other potential chemical stressors to the system are pH and mill reagents used at the mine. Because sulphidic materials are associated with the ore body, there is potential for acid rock drainage (ARD) to develop in areas where mine waste is deposited. Oxidation of sulphidic materials can lead to the generation of sulphuric acid. This acid may directly affect aquatic life by lowering ambient water pH and may also indirectly cause effects by mobilising metals. The upper reaches of the Ok Tedi/Fly river system are well buffered (mean alkalinity in Reach 2 is 348 mg/L), thus limiting the potential for the development of ARD conditions. In lower reaches, the alkalinity drops significantly, increasing the potential risk of ARD conditions developing in downstream portions of the system.

A number of chemical reagents are used in the milling process to separate copper from the parent material. The environmental fate of most of the chemicals is such that their persistence in the environment is too short to cause effects on aquatic life (Hawley 1972). The one exception to this may be xanthate compounds that are used as collectors in the milling process. Xanthates can be relatively stable under alkaline conditions. In addition to their own inherent toxicity, xanthates have been demonstrated to significantly enhance the bioavailability of metals (Block and Wicklund Glynn 1992; Borg et al. 1988; Gottofrey et al. 1988)

2.4.3.2 Sediments

As discussed above, the predominant fate of many chemicals released into surface waters is adsorption to sediment. Sediments may remain within the river system, discharge into numerous off-river water bodies, or flow with flood waters onto the floodplains adjoining the river. This section discusses the fate of chemical constituents once they reach the sediment. As in the section above, the chemicals are divided into three general classes.

Divalent Metals

Divalent metals bind to sediments by adsorbing to clay and other mineral surfaces, ion exchange, co-precipitation with hydrous iron or manganese oxides, and incorporation into cationic lattice sites in crystalline sediments (Rai et al. 1984; Callahan et al. 1979). Adsorption/desorption reactions are highly pH-dependent; as pH decreases, divalent metals are mobilised and released (Campbell and Lewis 1988; Eisler 1988).

Because the water column contains lower concentrations of particulates, organic carbon, and iron and manganese oxides than sediment, divalent metals are generally much more bioavailable in water than in sediment. Researchers have shown that dissolved divalent metals in sediment pore water represent the bioavailable metals in sediment (e.g., Di Toro et al. 1990; Ankley et al. 1991; Carlson et al. 1991; Di Toro et al. 1992; Casas and Crecelius 1994). More specifically, bioavailability can be measured by comparing the concentration of simultaneously extracted metals (SEM) with the concentration of acid-volatile sulphide (AVS). AVS is the sulphide fraction that is available to complex with dissolved metals (Di Toro et al. 1990, 1992; Casas and Crecelius 1994). When the molar concentration of these sulphides exceed the SEM concentration, these metals are rendered nonbioavailable and nontoxic. This relationship has been shown to be applicable to cadmium, copper, lead, nickel, and zinc (Di Toro et al. 1990, 1992; U.S. EPA 1994). It should be emphasised that SEM>AVS does not predict toxicity (e.g., Hare et al. 1994) and that SEM<AVS only predicts lack of toxicity for the metals presented above because other factors could result in the sediments being toxic (O'Connor et al. 1998). Additional studies have indicated that AVS:SEM measurements be interpreted cautiously as AVS can be quite variable with sediment depth and season (van den Berg et al. 1998).

Arsenic, Mercury, and Selenium

Similar to divalent metals, arsenic, mercury, and selenium tend to accumulate in sediment via adsorption to particles. A key difference between these three metals/metalloids compared to divalent metals, however, is their ability to be transformed into organic forms. For example, arsenic is biomethylated by microbial processes (Burton 1992; Masscheleyn et al. 1991) and mercury can be transformed between elemental, ionic, and methylated forms (i.e., monomethylmercury and dimethylmercury) by biotic and abiotic processes in anaerobic and aerobic environments (Mason et al. 1995a,b; Fitzgerald et al. 1994; Weber 1993; D'Itri 1993). The highest rates of methylation occur biotically in anaerobic sediments (Mason et al. 1995a,b; Rudd et al. 1983; Compeau and Bartha 1984). Microorganisms will transform inorganic selenides to organic selenides such as seleno-amino acids (e.g., selenomethionine, selenocysteine) and methylated selenium compounds (e.g., dimethyl selenide, dimethyl diselenide).

The formation of organic arsenic, mercury, and selenium compounds represents a significant source for remobilization of these metals/metalloids from sediment. These compounds are more bioavailable to aquatic organisms, and typically more toxic than their inorganic forms.

Other Chemical Stressors

The fate of other metals and metalloids in sediment is similar to the divalent metals discussed above (i.e., they will tend to bind to various particle surfaces or ions). Although some of the other metals can be converted to organic forms, these are not considered very important in the environment because they are present only at extremely low levels or they are not very toxic.

2.4.3.3 Soils

The environmental fate and transport of heavy metals and metalloid in soils have been reviewed by several authors; studied metals include Cu (CCME 1997a), As, Cd, Cr, Cu, Pb, Hg and Zn (CCME 1997b), Ag, Mo, Ni and Se (Alloway 1995) and Al, Fe, Mn (Lepp 1981a). These reviews provide the basis for the discussion of the potential chemical speciation and environmental fate of SOPCs in mine waste.

Divalent Metals

Copper is strongly adsorbed to soil particles and therefore has very little mobility relative to other trace metals (Alloway 1990). Soil types have finite holding capacities for copper ions and leaching can occur when the copper levels applied exceed this capacity (Adriano 1986). Soil factors that influence the availability of copper in soils are pH, cation exchange capacity (CEC), organic matter content, presence of oxides of iron, manganese, and aluminium, and reduction-oxidation potential (Adriano 1986; Slooff et al. 1989). Adriano (1986) showed that the capacity of soil to adsorb copper increased with increasing pH with a maximum holding capacity at neutral to slightly alkaline conditions (pH 6.7-7.8). Furthermore, soils with alkaline conditions tend to favour precipitation of copper; thus, copper is more mobile under acidic than alkaline conditions (CCME 1997a). In general, the higher the CEC, the greater the amount of copper it can adsorb (Adriano 1986). Soils with a high CEC have the ability to remove trace metal cations from the soil solution (Fuller 1977).

Copper has a very high affinity for organic matter and is more strongly bound than other trace elements (Lepp 1981b; Elsokkary and Lag 1978; Alloway 1990; Nriagu 1979; Slooff et al. 1989). Copper found in soil solution is often bound to dissolved organic matter (Lepp 1981b; CCME 1997a) but will release in an ionic form under strongly oxidising conditions or through microbial degradation of the organic matter (Fuller 1977). Copper is specifically adsorbed by iron, aluminium and manganese oxides (Alloway and Jackson 1991). With the possible exception of lead, copper is the most strongly adsorbed of all the divalent transition and trace metals on iron and aluminium oxides and oxyhydroxides (Adriano 1986).

A variety of factors influence the mobility of cadmium in soils including soil pH and salinity, soil type, particle size, content of metal oxides, hydroxides and oxyhydroxides, and organic matter content, being probably the most important (CCME 1997b). Numerous studies have identified soil pH as an important factor in influencing the mobility of cadmium in soil (Chanmugathas and Bollag 1987; Christensen 1989; Eriksson 1989; Lodenius and Autio 1989), and most studies indicate that significant movement of cadmium within the soil matrix and into other media is likely to occur under acidic conditions (CCME 1997b). Many studies have shown that clay minerals (Christensen 1984a, b; Inskeep and Baham 1983; McBride et al. 1981), metal oxides, hydroxides, and oxyhydroxides (Benjamin and Leckie 1981a, b; Bruemmer et al. 1988; Fu et al. 1991), and organic matter (Blume and Bruemmer 1991) are involved in the immobilisation of cadmium in soils. However, the presence of high concentrations of dissolved organic matter in soil leachates can also enhance cadmium mobility (Bollag and Czaban 1989; Christensen 1989).

Iron and manganese are essential plant nutrients. Iron and manganese oxides play an important role in the soil in fixing trace elements such as cobalt, copper, zinc, lead and nickel (Norrish 1975). The association of these elements with iron and manganese has important implications for plant growth. Studies have shown that fixation of elements by iron and manganese is rapid and results in the elements being unavailable to plants. The availability of manganese to plants is dependent on oxidising conditions and pH. Under acidic conditions, manganese is sufficiently soluble, but Mn deficiencies may occur with a pH of 7 to 8.

Lead has a relatively low solubility on soils (Cross and Taylor, 1996). Several factors influence the mobility and bioavailability of lead: pH, soil texture (especially clay content), and organic matter content (CCME 1997b). The Pb (II) ion is readily bound by soil organic matter and iron oxyhydroxides. In anaerobic soils, the reduction of SO_4^{2-} to S^{2-} leads to the formation of lead sulphide (PbS), a very low-solubility, low-reactivity lead species (CCME 1997b). Metallic lead has relatively low solubility and can be oxidised in soils to form PbO, which may then be dissolved or transformed into a more stable compound. In aerobic soils, weathering of soluble lead compounds may result in the formation of more stable compounds (Lindsay 1979).

Molybdenum is mainly associated with hydrous iron and aluminium oxides, and with organic substances. Molybdenum has been shown to be mobilised as anions by aerobically decomposing plant material with subsequent fixation by organic-colloidal complexes at low pH (Bloomfield and Kelso 1973). Anaerobic conditions result in the persistence of the anionic form. Molybdate is the dominant form of Molybdenum taken up by roots from soil solution (Edwards et al. 1995).

In general, nickel may be found in sulphide and oxide ores. Nickel can occur in the environment in a number of oxidation states, although only nickel (II) is stable over the wide range of pH and redox conditions found in the soil environment (McGrath 1995). Nickel has a relatively simple soil chemistry based on the nickel (II) divalent metal ion. Nickel solubility increases with decreasing pH, while other factors such as clay content and the amount of hydrous iron and manganese oxides in the soil are of secondary importance (Anderson and Christensen 1988). The mobility of nickel increases as the soil pH and CEC decrease. On the basis of thermodynamic stability, nickel ferrite ($NiFe_2O_4$) is the most probable solid phase that can precipitate in soils (Sadiq and Enfield 1984a,b). Where the soil environment is acid and reducing, the sulphides of nickel are likely to control the concentration of nickel in the soil solution. The hydroxy-complex Ni(OH) (I) and Ni (II) ions are the most likely major forms of nickel in the soil solution above pH 8, whilst in acid soils Ni (II), $NiSO_4$, and $NiHPO_4$ are important although the relative proportions would depend on the levels of SO_4^{2-} and PO_4^{3-} (CCME 1997b).

Silver is a rare element that is distributed in the earth's crust at a concentration of approximately 0.1 mg/kg. Silver can occur as native silver, in mineral form, and as complex sulphides. Oxidation states of silver include Ag (I), Ag (II) and Ag (III). Silver exhibits low solubility of most of its compounds and high toxicity of the soluble fraction (Cooper and Jolly 1970). Silver sulphide (Ag_2S) is relatively soluble but has a low ionisation potential. In highly reduced soils, silver will precipitate in sulphide minerals and all silver halide minerals are sufficiently insoluble to be considered stable. Behaviour of silver in soils is strongly influenced by pH and redox potential conditions, and by interactions with soil organic matter. In field soils, silver tends to accumulate in the surface, organic-rich horizons (Presant and Tupper 1963; Romney et al. 1979). In the soil, silver is persistent and leaching is limited relative to other metals (Cameron 1973; Khan et al. 1982). The solubility of silver in soil increases with soil humus decomposition and reduced pH (Cameron 1973; Khan et al. 1982). Humic acids and organic matter probably limit adsorption and retention of silver, and excess silver in soils of low organic matter content may be more toxic than in soils of higher organic matter content (Kabata-Pendias and Pendias 1992).

Zinc is highly reactive in soils, so that in addition to inorganic Zn (II), zinc is present as part of both soluble and insoluble organic compounds (CCME 1997b). Zinc can also be adsorbed to clay minerals and metallic oxides and may be present within primary minerals of the soil parent material (Sachdev et al. 1992). The concentration of zinc in soil solution is dependent upon the amount of zinc present in the soil, solubility of the particular zinc compound, and the extent of adsorption (CCME 1997b). Zinc compounds vary significantly in solubility; zinc sulphate is readily soluble in soil solution, while zinc oxide has relatively low solubility. Zinc may be adsorbed to clay minerals and may also form stable compounds with soil organic matter, hydroxides, oxides and carbonates (CCME 1997b). Soil pH has been identified in many studies as one of the main factors affecting zinc mobility and sorption in soils (Davis-Carter and Shuman 1993; Duquette and Hendershot 1990; Evans 1989; Shuman 1975). Zinc becomes more soluble as pH decreases, therefore zinc is more mobile and increasingly available to organisms in low pH environments, especially below pH 5 (Duquette and Hendershot 1990).

Arsenic, Mercury, and Selenium

Many factors control the fate and behaviour of arsenic in soil and ultimately its bioavailability (CCME 1997b). It is generally accepted that only soluble arsenic is available for plant root uptake (CCME 1997b). The solubility and speciation of arsenic in solution are primarily determined by pH and reduction/oxidation potential (CCME 1997b). The solubility of arsenic may determine the amount of arsenic ultimately available within a system, while speciation of arsenic determines the behaviour, and to a large extent, toxicity of arsenic (CCME 1997b).

Arsenic in soil-water environments can be present in at least four chemical forms: arsenate (As [V]), arsenite (As [III]), monomethyl arsenic acid and dimethyl arsenic acid (Marin et al. 1992). Arsenic is subject to chemically and/or microbiologically mediated oxidation-reduction and methylation reactions in soils (Marin et al. 1992). Generally, in oxidising conditions or aerobic soils, inorganic arsenic is predominantly present as arsenate, AsO_4^{-3} (CCME 1997b). Under reducing conditions, as in anaerobic waterlogged soils, arsenic is primarily present as arsenite, AsO_3^{-3}. Under strongly reduced conditions elemental arsenic and arsine (AsH_3) can exist (CCME 1997b).

One of the most important processes influencing the behaviour and bioavailability of arsenic in soil is its ability to sorb onto particulates (CCME 1997b). Factors governing the sorption of arsenic in soil include pH and the amount of clay, iron, aluminium, calcium, and phosphorus present. Arsenic is strongly adsorbed by clay minerals (Dickens and Hiltbold 1967; Slooff et al. 1990), and electropositive hydroxides such as aluminium, iron, and calcium, that coat clay particles or react with cations in the soil solution (Jacobs et al. 1970).

The major soil factors that determine the fate and behaviour of mercury are pH, organic matter and clay content, redox potential, cation exchange capacity (CEC), aeration, and texture. The major processes that determine the mobility and distribution of mercury in the terrestrial environment are adsorption, chemical reactions, leaching, volatilisation, photolysis and biodegradation. These processes are dependent on the soil factors mentioned above.

In soils, mercury occurs mainly in the Hg (0) and Hg (II) oxidation states. Depending on redox conditions, the dimeric ion Hg_2 (II) may also be encountered. Mercury speciation in soils also depends on the pH and the concentration of chloride ions. Under natural conditions, most of the Hg (II) in the soil is either bound in the soil minerals or adsorbed onto organic or inorganic solids with only a very small portion present in the soil solution (Steinnes l995).

Adsorption is the dominant process determining the fate of mercury in the terrestrial environment (Hogg et al. 1978). It depends on the chemical form of mercury, soil pH, colloids, CEC and redox potential (Hogg et al. 1978; Kabata-Pendias and Pendias 1992). Adsorption is increased by the presence of organic matter due to mercury complexation with humic and fulvic acids, and therefore, adsorption is greater in surface horizons due to high humus content (Lodenius et al. 1987).

Mercury undergoes methylation by aerobic and anaerobic bacteria (NRCC 1979; WHO 1991). Methylmercury species are relatively more mobile, bioavailable and highly toxic (Bigham and Henry 1993). Under reduced conditions, mercury and sulphide ions form HgS, an insoluble salt that is resistant to methylation. Under aerobic conditions, HgS is oxidised to the sulphate form $HgSO_4$, which can undergo methylation. Bacterial action can cause demethylation of methyl mercury compounds. The chloride concentration and pH determine the chemical form of monomethylmercuric ion complexes (NRCC 1979).

Selenium may occur in inorganic or organic forms in the soil. Inorganic oxidation states include II, III, IV and VI, all of which may be found under a variety of soil conditions (CCME 1997b). Redox potential, pH, soil organic matter and microbial transformations may affect the form of selenium in soil (Peterson et al. 1981). The inorganic oxidation state of selenium in soils is largely defined by pH and redox potential. In soils with a pH range of 4 to 9 and Eh range of -5 to 15, the stable inorganic oxidation states are:

- Se (VI) as selenate (SeO_4^{2-});

- Se (IV) as selenite ($HSeO^{3-}$) or biselenite (SeO_3^{2-}); and
- elemental Se (Se^0).

Geering et al. (1968) indicated that selenium solubility in soils was governed by the ferric oxide-selenite adsorption complex. Selenium may therefore concentrate in clay and iron hydroxide sediments (Peterson et al. 1981). Selenium geochemistry is controlled by iron, with which selenium is affiliated in both oxidising and reducing environments (Peterson et al. 1981).

In acidic (pH 4.5 - 6.5), aerated soils, selenium may bind with iron as ferric selenium forming low solubility compounds that are essentially unavailable to plants. In waterlogged, acidic soils, elemental selenium and selenide forms occur (Peterson et al. 1981), particularly in anoxic, deeper soils. In alkaline soils, selenium may be oxidised to the selenate ion (Peterson et al. 1981). Selenate predominates in well aerated, alkaline soils, such as those in arid areas. In wetter climates or waterlogged soils, the formation of selenate is likely to be less (Ohlendorf 1989). Selenate is the most mobile and soluble of the inorganic species of selenium. In neutral or acid soils, selenite or biselenite will predominate in the soil solution, and will be relatively less available for plants due to adsorption onto clays and hydrous oxides.

Selenium may also occur in organoselenium forms, such as organoselenides, in soils. Organic forms of selenium are considered less available than inorganic forms, particularly selenate. Selenate may undergo microbial reduction in anaerobic soils to the relatively lower solubility forms of selenium including elemental Se and elemental forms of organic Se (Weres et al. 1989).

<u>Other Chemical Stressors</u>

Aluminium is third in order of abundance in the earth's crust but is the most common metallic element (Petersen and Girling 1981). Soils, on average, contain approximately 71,000 mg/kg aluminium (Bowen 1966). Aluminium is highly reactive in nature, and unlikely to be found in an uncombined state (Petersen and Girling 1981). Soluble aluminium in acid soils may cause adverse effects to plants.

The degree to which Cr (III) can interact with other soil constituents is limited by the fact that most Cr (III) is present in the form of insoluble chromium oxide precipitates (CCME 1997b). Thus, Cr (III) is relatively stable in most soils (Kabata-Pendias and Pendias 1984), although oxidation of Cr (III) to Cr (VI) can occur under specific environmental conditions. Factors influencing the rate of chromium oxidation include soil pH, Cr (III) concentration, presence of competing metal ions, availability of manganese oxides, presence of chelating agents (i.e., low molecular weight organic compounds), and soil water activity (CCME 1997b).

Cr (III) oxidation is favoured under acidic conditions (Bartlett 1986; Bartlett and James 1979; Fendorf et al. 1992). This behaviour is attributable to increased solubility of Cr (III) at lower pH that enables increased contact with the oxidising agent (Bartlett 1991). Cr (III) must be in a mobile form to undergo oxidation on the surfaces of manganese oxides (Bartlett 1991). Aside from decreasing soil pH, Cr (III) solubility is enhanced by chelation to low molecular weight compounds such as citric or fulvic acids (Bartlett and James 1988).

Factors influencing the reduction of Cr (VI) to Cr (III) in soil include soil pH, the presence of electron donors such as organic matter or ferrous ions, and soil oxygen levels (CCME 1997b). Many studies have shown that that Cr (VI) reduction increases with decreasing soil pH (Bartlett 1991; Bartlett and Kimble 1976; Bloomfield and Pruden 1980; Eary and Rai 1991). Soil pH effects the degree of positive and negative charge on the surfaces of soil colloids, thus directly influencing the availability of electron donors (Bartlett and James 1988). Rai et al. (1989) concluded that acidic soil solutions enhance the release of Fe (II) ions from soil minerals, which increased the reduction of Cr (VI).

Cr (III) is strongly adsorbed by clay particles, soil organic matter, metal oxyhydroxides and other negatively charged particles. Below pH 4, Cr (III) is strongly adsorbed by both kaolinite and montmorillonite clays. Between pH 4 and 5 the combination of adsorption and precipitation renders this species immobile in most soils (Jaworski 1985; NRCC 1976). Since clay surfaces become more negatively charged with increasing pH, Cr (III) adsorption by clay minerals increases with increasing soil pH. Although Cr (VI) is not readily adsorbed to most surfaces, it is adsorbed by clay minerals that possess exposed inorganic hydroxyl groups, including iron and aluminium oxides (Rai et al. 1989; Zachara et al. 1989). Cr (VI) adsorption increases with decreasing pH as a result of protonation of the surface hydroxyl sites.

2.4.3.4 Tissue

<u>Divalent Metals</u>

Divalent metals generally do not bioaccumulate extensively in the tissue of fish, birds, and mammals. The literature demonstrates that bioaccumulation of divalent metals is primarily due to uptake through the gills or surface membranes of aquatic life (i.e., fish, aquatic invertebrates, and plants). Most metals bioaccumulate in the tissue of aquatic organisms; however, the extent of bioaccumulation varies, depending on the metal and the organisms considered. Bioaccumulation in tissue can be estimated using bioconcentration factors (BCFs) and bioaccumulation factors (BAFs). A BCF is the ratio of the amount of a chemical accumulated in the tissue to that in the water (i.e., chemicals accumulated via gills and epithelial tissue). Whereas, a BAF is a measure of the process whereby chemicals enter aquatic organisms through the gills and epithelial tissue directly from the water and from food items. Therefore, BAFs incorporate uptake both through gills as well as via food.

Metal bioaccumulation is species-specific and largely a function of the metal regulatory strategy utilised by the organism. For example, BAFs for zinc in eastern oysters (*Crassostrea virginica*) range from 17,640 to 23,820, while for mummichogs (*Fundulus heteroclitus*) BCFs range from 4.5 to 41 (U.S. EPA 1987b). In another example, BCFs for copper in Asiatic clams (*Corbicula fluminea*) range from 17,700 to 22,600 while for fathead minnows (*Pimephales promelas*) and bluegills (*Lepomis macrochirus*), BCFs range from 1 to 290 (U.S. EPA 1985a).

Although water is the typical exposure pathway for aquatic organisms, uptake of metals via food and particulates can be important particularly in systems with high metals loads like the Ok Tedi/Fly River system. Metals can be absorbed through the gut from ingested items (Dallinger et al. 1987). In a study at a metal-contaminated stream in the United States, brown trout (*Salmo trutta*) accumulated more copper and lead when exposed via both food and water versus water alone (Woodward et al. 1995). The brown trout accumulated approximately 25 percent of the total copper burden via food (Woodward et al. 1995). Both brown trout and rainbow trout (*Oncorhynchus mykiss*) also accumulated arsenic when dietary uptake was the only source (Woodward et al. 1995). Uptake of particulate metals by fish has not been well documented. However, considering the high levels of particulate metals in the Ok Tedi/Fly River system, this pathway needs evaluation.

Divalent metals also bioaccumulate in terrestrial food chains; however, their BAFs are generally low. For example, cadmium, chromium, copper, nickel, and zinc were not found to accumulate significantly, when compared to controls, in mice (*Peromyscus leucopus*) fed a diet of pellets that were made from rye grass grown in municipal sludge that had high concentrations of these metals (Woodyard and Haufler 1991). Field-based BAFs for copper and cadmium in plant-granivore and plant-frugivore food chains are less than one (Hunter and Johnson 1982). In another field study, arsenic, cadmium, copper, lead, and zinc were not bioaccumulated at high rates in a plant-herbivore food chain; the bioavailabilities were less than 0.2 percent for the herbivore internal organs and less than 0.1 percent for their whole bodies (Pascoe et al. 1994).

Although divalent metals generally do not bioaccumulate strongly in tissue, the metal in the tissue may be bioavailable when consumed by a bird or mammal. For example, metals bound to sulphur-based proteins are bioavailable (Henry et al. 1992). The bioavailability of manganese to lambs was lowest when given as MnO and highest as a metal- amino acid complex (Mn-methionine) (Henry et al. 1992). Similarly, the bioavailability of cadmium to mice was lowest when given as $CdCl_2$ and highest when incorporated into oyster tissue (Sullivan et al. 1984).

Arsenic, Mercury, and Selenium

Arsenic is an example of a metalloid that can be bioaccumulated from water by many organisms, but there is no evidence that it is biomagnified through aquatic food chains (Woolson 1975; Hallacher et al. 1985; Hood 1985). BCFs for freshwater invertebrates and fish are relatively low. BCFs determined experimentally for arsenic were 17 or less for inorganic As (III), 6 or less for inorganic As (V), and 9 or less for organoarsenicals (U.S. EPA 1985b).

Mercury and selenium generally do bioaccumulate in tissue of fish, birds and mammals. For mercury and selenium, the dietary component is generally more important than the water component (Biddinger and Gloss 1984; Dallinger et al. 1987). For example, it has been found that dietary selenium can have significant effects in some species of aquatic life (U.S. EPA 1988b). Factors that affect bioaccumulation include the ability of most species to detoxify or sequester some of the accumulated metal. In a review on the importance of trophic transfer in accumulation of chemicals in aquatic organisms (Biddinger and Gloss 1984), no metals other than methyl mercury and selenium were found to have the potential for food chain biomagnification.

Other Chemical Stressors

Other chemical stressors, including calcium, magnesium, sodium, and sulphate, generally do not bioaccumulate in fish, bird or mammal tissue. These metals are regulated by common metabolic mechanisms and are often essential for life (Rand and Petrocelli 1985; Beeby 1991). Sodium for example, is a highly regulated ion and is needed for proper neuron function (Kuffler and Nicholls 1976) and water uptake (Ganong 1983). All of these common ions are toxic only at very high concentrations.

2.5 RESOURCES TO BE PROTECTED

The following section provides a description of the resources to be protected. These resources include human health and various components of the terrestrial and aquatic ecosystems. From a societal perspective, significant risks to human health are inherently unacceptable. Accordingly, defining this resource is relatively straightforward as described in Section 2.5.1.

Defining ecological resources to be protected requires more analysis because critical components of the ecosystem are not always intuitively obvious, and different levels of ecological organisation must be considered. The process of defining "general assessment endpoints" for ecological resources provides an objective and thorough means of identifying key components and ensuring their protection. This process is described in Section 2.5.2.

2.5.1 Human Populations and Water-Dependent Human Uses of the Ok Tedi/Fly River System

The section below describes the manner in which human health may be affected by mine-related stressors and describes the various ways these stressors may adversely affect human populations in the study area.

Risks to human health may occur due to exposures to metals and other inorganic substances. These adverse effects may occur when people are exposed chronically (long-term) to mine wastes or environmental media impacted by them. These exposures occur when people incidentally drink, bathe, or swim in river water, ingest sediments affected by mine wastes, or eat food harvested from the Ok Tedi or Fly River. The extent to which people may be affected by chemical stressors from mine tailings is determined by the intensity with which water, food, or other environmental media affected by mine wastes are used. For example, drinking water may consist of surface, groundwater, or rainwater. The degree to which each of these media are used for drinking water and the effect of tailings on each of these water sources determines the intake of chemical stressors via the drinking water pathway.

An additional source of potential human health effects is the potential indirect effect on local food resources. Aquatic species or wildlife may be exposed to mine wastes transported by the Ok Tedi and Fly rivers. If these exposures result in population-level effects to the exposed aquatic or terrestrial species, any resulting population declines will affect the availability of foods to the extent these species are used as a food resource. In addition to these food resource effects, there is the potential for adverse cultural effects. In some New Guinea populations, game is used for ritual consumption and ceremonial distribution (Bulmer 1968). In some situations food crops have been reported to have a ceremonial purpose among some south-central New Guinea peoples (Blackwood 1940). To the extent that any aquatic or wildlife species are used for cultural purposes at villages in the Ok Tedi and Fly River basins, a decline in these species could have an adverse cultural impact. The potential for cultural impacts will not be evaluated directly in the HERA; rather, these effects will be assessed in the social risk assessment conducted as part of the overall MWMRA.

2.5.2 Ecological Values

2.5.2.1 General Assessment Endpoint Approach

In ecological risk assessments, assessment endpoints (values to be protected) are identified as a basis for evaluating ecological risk (U.S. EPA 1992b). Because ecological values at all levels of organisation may be adversely affected by stressors, the process of determining ecological values can be difficult. This is particularly true for large complex sites such as the Ok Tedi/Fly River basins where the ecological resources are diverse but poorly known and natural disturbances such as floods, droughts, and fire exert substantial influence on ecosystem structure and function.

This section describes the comprehensive, systematic, and objective process for identifying ecological values to be protected from physical, chemical, and biological stressors directly or indirectly resulting from mine waste discharge. These general assessment endpoints (GAEs) encompass ecological and human use values at all levels of ecological organisation, thus providing a basis for determining site-specific assessment endpoints (i.e., those endpoints that may be, or are currently being, adversely affected by mine-derived stressors).

While there is currently no guidance concerning how to systematically identify ecological values, U.S. EPA (1998a) provides three criteria for selecting assessment endpoints:

- ecological relevance
- relevance to management goals
- susceptibility to known or potential stressors

Ecologically relevant endpoints are those that reflect important ecological characteristics and are functionally related to other endpoints (U.S. EPA 1992b, 1998a). Endpoints based on ecological values that people care about (i.e., societal values) have management relevance and are, therefore, more likely to be used in risk management decisions.

In determining ecological values to be protected, the ultimate value to be protected is a healthy sustainable ecosystem. Ecological relevance, as used here, refers to the properties necessary for normal ecosystem function.

Process Overview

The process of identifying GAEs occurs in two parts. Part one is performed to identify values from an ecological perspective; part two considers the human values associated with the resources of the ecosystem under evaluation.

The process consists of five steps:

1) identification of values common to all ecosystems
2) identification of functional ecosystem components
3) development of a functional food web
4) determination of attributes of the functional components of the ecosystem
5) description of ecologically relevant GAEs

A more detailed description of the GAE process is presented in Reagan (In Press).

Once GAEs have been determined, values relevant to management goals (i.e., societal values) are described as part two of the GAE process. Subsequent sections describe the GAE development process for terrestrial and aquatic ecosystems.

Common Ecological Values

For purposes of this ecological risk assessment, common ecological values have been defined as:

- Biological diversity (biodiversity) - This property describes ecological structure in terms of components and can include species, community/habitat, and genetic diversity. From a generic perspective, biological diversity is usually considered most relevant in the context of species diversity (i.e., the variety and abundance of species) (Norton 1987; Magurran 1988) that is typical of particular ecosystems. Communities that are more diverse are not necessarily more relevant than less diverse communities (Paine 1966), but communities in disturbed ecosystems are usually less diverse than those in comparable but undisturbed ecosystems.

- Functional integrity - For the purpose of defining assessment endpoints, functional integrity is defined somewhat narrowly as the pattern of interactions among components of the ecosystem. This allows discrimination between species composition in the ecosystem (e.g., biodiversity) and the functional interactions among components. Thus, assessment endpoints can distinguish patterns such as trophic structure or habitat relationships among specific species or functional guilds in addition to capturing biological diversity. In practice, to assess functional integrity, factors such as food chain length, connectivity, degree of omnivory, extent of reciprocal predation (food loops), and subweb organisation can be evaluated (Reagan et al. 1996).

  Functional integrity is a valued attribute because it connotes an intact system — one in which there is no missing link that would result in structural or functional imbalances that render the entire system more vulnerable (less resilient) to perturbation. An ecosystem with integrity has as its primary link a set of living organisms within populations of different taxa, some of which modify their surroundings primarily by altering the system's abiotic features and secondarily through effects on other biota that in turn alter features through the use of other biota as food (Reiger 1993).

- Energy and nutrient dynamics (cycling and transport processes) - The flow rates and patterns of nutrient and energy processing in a given ecosystem are critical for maintaining populations of indigenous species at levels characteristic of that ecosystem. For an ecosystem to operate unimpaired, nutrient and energy flow through the system should also remain unimpaired or unaltered by anthropogenic activity. Disruption of nutrient and energy flow rates (e.g. by nutrient enrichment or chemical contamination) can lead to accumulation of detritus, reduction of primary productivity, or loss of top predators.

These common ecological values express the basic considerations of ecosystem structure and function using fundamental ecological concepts expressed in terms that can be evaluated in the risk assessment (e.g., as characteristics that can be measured). They are relevant to all ecosystems (Kormondy 1969, Brewer 1979, Odum 1993). All other ecologically relevant issues are subsets of these basic aspects of ecological organisation. Figure 4 illustrates the hierarchical relationship among ecological values for the terrestrial ecosystem of the Ok Tedi/Fly River basin.

For purposes of the SLRA, some of the measurement endpoints for the above ecosystem-level values were evaluated directly (e.g., screening values for protection of aquatic ecosystems consider biological diversity and ecological function). However, for some endpoints this was not possible. In these instances literature values for protection of individuals and populations were used to compensate for the lack of site-specific data and to be adequately conservative. It was assumed that protection at these levels would afford adequate protection at higher levels of ecological organisation.

2.5.2.2 Terrestrial Ecosystem

The terrestrial ecosystem includes the plant communities bordering the channel of the Ok Tedi/Fly Rivers from the Ok Tedi Mine to the Fly River Estuary. Extensive areas of mangrove occur throughout the Fly River Estuary. These vegetation communities are values to be protected because of their contribution to biological diversity, their fundamental role in primary productivity, and their function as food and habitat for the animal community.

The ultimate ecological value to protected is a sustainable healthy ecosystem; however, a variety of other ecological values must also be considered. The process of identifying these values, beginning at the ecosystem level and progressing to lower levels of ecological organisation, is described in this section.

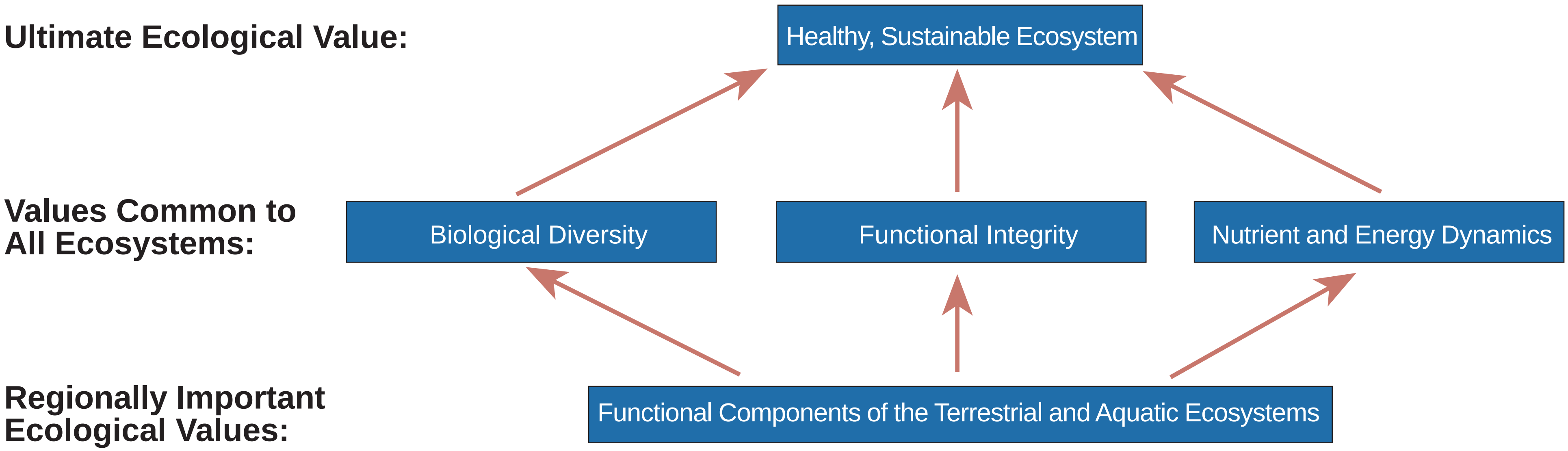


Figure 4.
Hierarchy of Ecological Values for the Terrestrial and Aquatic Ecosystems

Functional Components of the Terrestrial Ecosystem

Once common ecosystem values have been identified, values of the regional ecosystem potentially affected by stressors are determined. To determine these values for the regional ecosystem, the principal functional components of the ecosystems are first identified. Because food webs provide essential structural organisation in ecosystems (Gallopin 1972), and because all organisms in an ecosystem are part of the food web, this concept is used to identify basic functional components of regional ecosystem.

Food webs typically contain three basic trophic categories:

- Producers - organisms that manufacture food from inorganic compounds by photosynthesis or chemosynthesis (e.g., green plants);
- Consumers - organisms that ingest other organisms (e.g., animals that consume plants or other animals); and
- Decomposers - organisms that derive their nourishment from dead organic matter (e.g., fungi and bacteria).

These categories are based on the broad interrelationships among groups of organisms but do not describe the many ways in which these interactions may occur. Organisms that obtain their food in a functionally similar way constitute a feeding guild. Food webs based on feeding guilds facilitate the identification of critical ecosystem functions performed by members of each guild and the interrelationships among guilds that may affect other ecosystem properties.

The three fundamental food web categories were divided into functional groups (components), based on a general knowledge of the species present in regional ecosystems and from OTML reports. Functional components for the terrestrial ecosystem are presented in Table 2.

These categories can be further subdivided, based on specific knowledge of the importance of some groups. For example, bats and butterflies pollinate different plants, and some relationships are species-specific. To the extent that these are known to be important in overall ecosystem function, they can be incorporated into the determination of assessment endpoints for the baseline risk assessment.

While all tropical rain forests have similar functional components, the plants and animals of the terrestrial ecosystems of New Guinea have close affinities to Australia but differ from the rest of the world. Tropical rain forests in the Americas, Africa, and Southeast Asia typically have monkeys as prominent arboreal herbivores and cats (e.g., tigers) as top predators; however, neither group occurs in New Guinea, which has been isolated from these regions. Tree kangaroos are the ecological equivalent of monkeys, and the largest predators in the forest are pythons and wild dogs (introduced in New Guinea about 2,000 years ago). Information on the species present in the region and on their habitats, food habits, and foraging strategies were obtained primarily from Mackay (1976), Beehler et al. (1986), and Flannery (1995).

Table 2. Functional components of the terrestrial ecosystem of the Ok Tedi/Fly River systems.

Fundamental Category	Functional Component
Producer	Ground layer
	Woody shrubs
	Understory trees
	Canopy trees
	Lianas
	Epiphytes
	Mycorrhizal fungi (enhancers)
Consumers	Herbivores
	Terrestrial frugivores/granivores
	Arboreal frugivores
	Folivores
	Nectarivores
	Carnivores and Omnivores
	Intermediate and small predators
	Omnivores
	Top terrestrial predators
	Top arboreal predators
	Detritivores
Decomposers	Chemical decomposers

The Terrestrial Food Web

The functional components described in the previous section define the general range of feeding preferences and location (strata) in forest habitat. A food web based on these functional components is presented in Figure 5. The arrows in the food web diagram indicate the direction of energy flow and nutrients through the food web. Dashed lines indicate the recycling and flux of energy and nutrients as the result of decomposition processes. Organisms of different sizes are grouped in a single functional component, based on similarity in food and foraging mode. These groupings convey a possible impression that the smaller organisms in one functional group prey on the larger organisms; however, scale must be considered in determining specific feeding interactions. Such a diagram is also simplistic in that it does not take into account reciprocal predation (food loops) and the division of the food web into day and night compartments, such as those that occur in other tropical forest food webs (Reagan and Waide 1996). The functional elements provide a basic framework for determining assessment endpoints for the SLRA. Specialised considerations can be addressed in the subsequent detailed phase of the HERA.

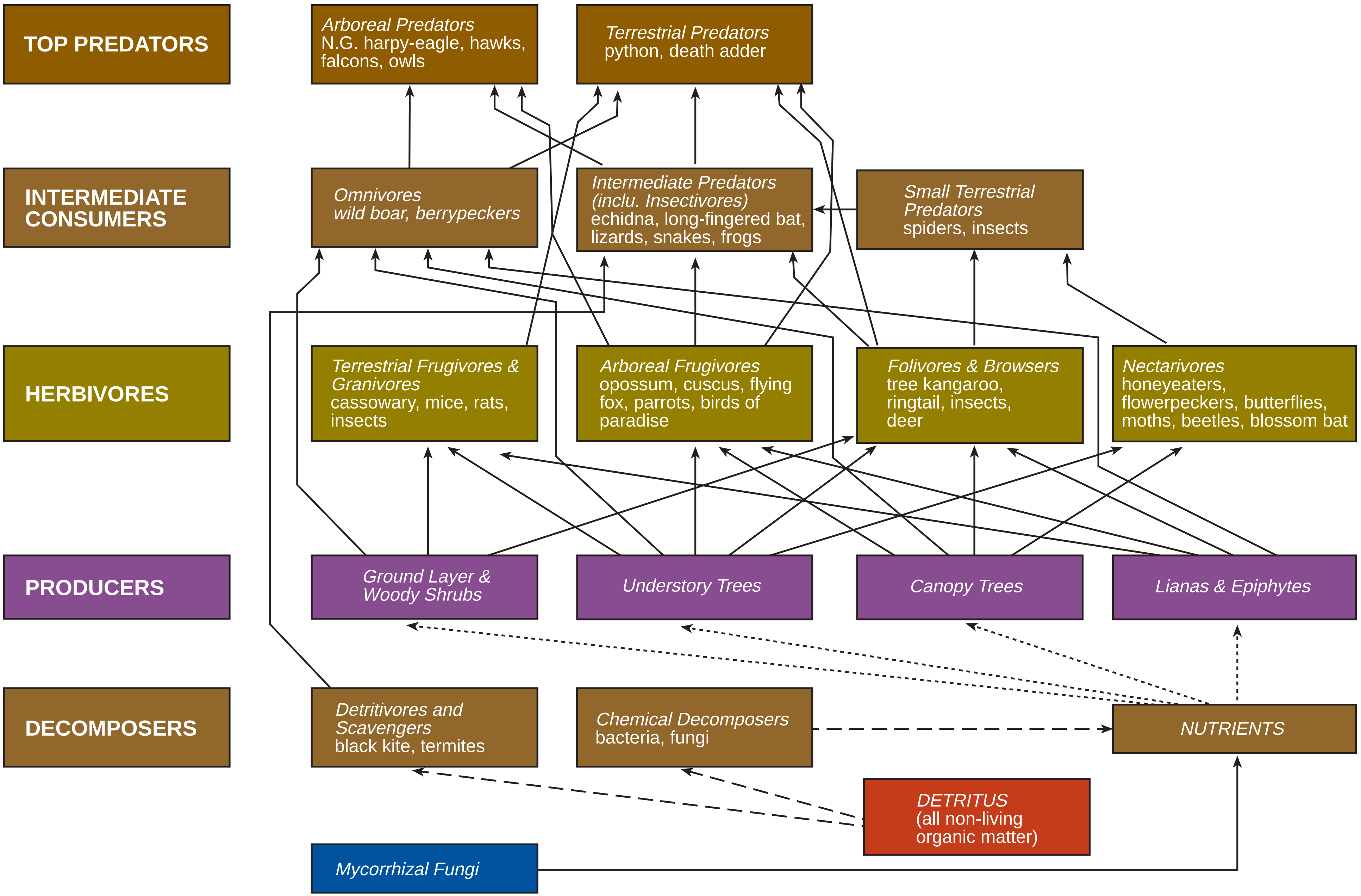


Figure 5.
Functional Food Web for Terrestrial Ecosystems of the Ok Tedi - Fly River Basin

Ecologically Relevant Attributes

While feeding relationships are relevant characteristics of each functional component of the terrestrial food web, each component may have additional attributes that define its overall ecological value. For many functional components, the non-trophic attributes are at least as important as their role in nutrient and energy transfer through the food web.

A preliminary list of ecologically relevant attributes for each functional subgroup is provided in Table 3. The list was developed from a general knowledge of tropical forest ecosystems and a superficial knowledge of the species present or likely to occur in the terrestrial ecosystem of the OTML study area.

Table 3 shows the general functional components of the terrestrial ecosystems of the Ok Tedi/Fly River basin. However, these components will not be individually evaluated in the SLRA. In the SLRA, physical and chemical stressors will be evaluated based on benchmark criteria that are designed to be protective of all levels of ecological specificity (e.g., biodiversity, herbaceous plants, and protected species), to the extent that relevant benchmark criteria are available. If soil concentrations for a given chemical stressor fall below that stressor's benchmark criterion for potential adverse effect, we assume that there is no risk to the terrestrial community from that stressor. Individual functional components will be assessed, as appropriate, in the DLRA, which follows the SLRA.

2.5.2.3 Aquatic Ecosystem

As in the terrestrial ecosystem, the aquatic ecosystems (freshwater and estuarine) can be divided into three fundamental categories: producers, consumers, and decomposers. The definitions for these categories in the terrestrial ecosystem also apply to the aquatic ecosystems. The functional components of these general categories are shown in Table 4. In general, the table is assumed to be relevant to both the freshwater and estuarine ecosystems; however, species that constitute each functional component will differ. A model food web for the freshwater system is presented in Figure 6 (Storey and Smith 1998). The estuary has not been sufficiently described to develop a food web. If this area requires further assessment in the DLRA, a food web will need to be developed.

Table 4 shows the general functional components of the aquatic ecosystems, but it should be noted that these components will not be individually evaluated in the SLRA. In the SLRA, chemical stressors will be evaluated based on water quality criteria that are designed to be protective of the entire aquatic community (i.e., water quality criteria are not available for individual components of the aquatic community). If surface water concentrations for a given stressor fall below that stressor's water quality criterion, there is assumed to be no risk to the aquatic community from that stressor. Individual functional components will be assessed as needed in the DLRA, which follows the SLRA.

Table 3. Ecologically relevant attributes of the functional components identified for the terrestrial food web of the Ok Tedi/Fly River systems.

	Ecological Attributes						
Functional Subgroups	Food	Habitat	Primary Production	Pollination	Seed Disp.	Decomp.	Control
Ground Layer Plants	X	X	X				
Woody Shrubs	X	X	X				
Understory Trees	X	X	X				
Canopy Trees	X	X	X				
Lianas	X	X	X				
Epiphytes	X	X	X				
Mycorrhizal Fungi			X				
Terrestrial Frugivores/Granivores	X				X		
Arboreal Frugivores	X				X		
Folivores	X						
Nectarivores	X			X			
Intermediate & Small Predators	X						
Omnivores	X						
Top Terr. & Arboreal Predators							X
Detritivores						X	
Chemical Decomposers						X	

Table 4. Functional components and their ecological attributes in the aquatic ecosystems.

Functional Category	Functional Component	Food	Habitat	Primary Production	Control	Decomposition
Producer	Algae	X		X		
	Aquatic vegetation	X	X	X		
	Terrestrial vegetation	X	X	X		
Consumer	Aquatic invertebrates	X				
	Terrestrial invertebrates	X				
	Aquatic vertebrates	X				
	Terrestrial vertebrates	X				
	Terrestrial arthropod-eating fish	X				
	Terrestrial vertebrate-eating fish	X				
	Omnivores	X				
	Top predators				X	
	Detritivores	X				
Decomposer	Bacteria					X

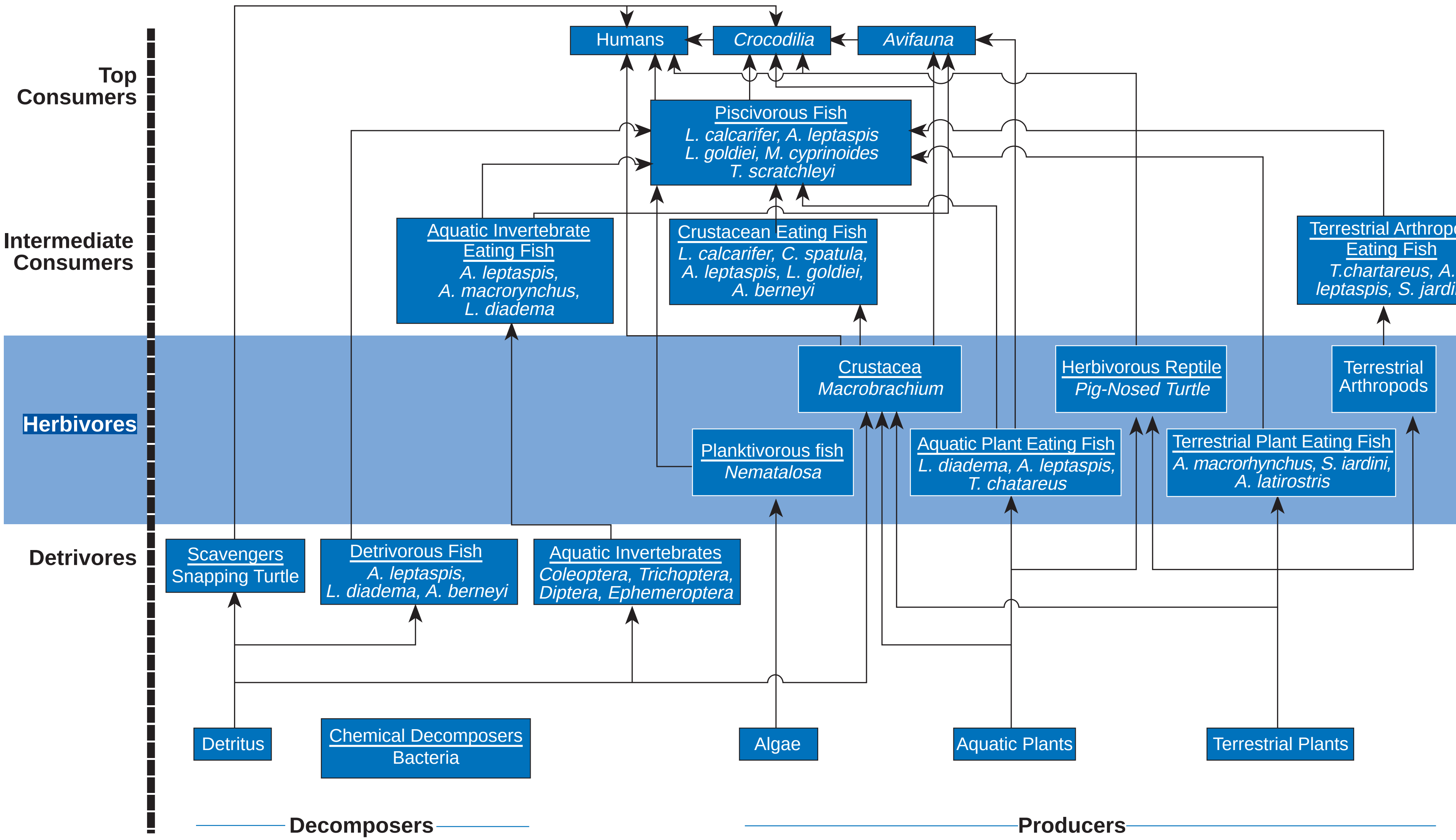


Figure 6.
Functional Food Web for Aquatic Ecosystems of the Ok Tedi - Fly River Basin

2.5.2.4 Ecologically Relevant General Assessment Endpoints

An endpoint is defined as a characteristic of an ecological component (Suter 1990). The attributes tables for terrestrial and aquatic ecosystems therefore provide the basis for describing the GAEs that are ecologically relevant for the study area. In descending order of hierarchical organisation these are:

- Healthy sustainable terrestrial and aquatic ecosystems;
- Biological diversity of terrestrial and aquatic ecosystems;
- Functional integrity of terrestrial and aquatic ecosystems;
- Energy and nutrient dynamics of terrestrial and aquatic ecosystems;
- Primary productivity, habitat, and food value of terrestrial plants;
- Primary production value in aquatic ecosystems;
- Mycorrhizal enhancement of primary production in terrestrial ecosystems;
- Terrestrial and arboreal frugivores and granivore food and seed dispersal;
- Folivore food value;
- Aquatic herbivore, omnivore, and carnivore value as food;
- Nectarivore food and pollination value;
- Intermediate and small predators and omnivore food value;
- Top predator control of ecosystem organisation;
- Detritivore and chemical decomposer roles in decomposition;
- Aquatic and terrestrial vegetation food, habitat, and primary production values in aquatic ecosystems;
- Detritivore food and decomposition value; and
- Bacterial decomposition value.

2.5.2.5 Societal Values

Ultimately, the effectiveness of an ecological risk assessment depends on how it improves the quality of management decisions. Management goals are inextricably related to the societal values of ecological resources. Therefore, the ecological risk assessment must consider both ecological and societal values to be effective.

Several wildlife species inhabiting the study area are hunted and eaten by local people. These include cuscus, tree-kangaroos, cassowaries, crocodiles, and turtles. Some species range throughout most of the habitats found in the study area, while others (e.g., *Dorcopsis* spp.) are confined to specific habitats and, therefore, occur in only a portion of the study area.

From a global perspective, many of the species inhabiting the study area are also valued because they are threatened with extinction, and because of larger concerns for loss of biological diversity. The IUCN (1996) lists 57 mammal, 31 bird, 10 reptile, 13 fish, and 11 invertebrate species as threatened in New Guinea. Several of these have been reported from the region or are likely to inhabit the study area (e.g., the Fly River grassbird).

2.5.3 Assessment Endpoints

The GAEs developed in the previous section provide a comprehensive summary of the ecological values for the terrestrial and aquatic ecosystems of the study area. These have been determined to ensure that values at all levels of ecological organisation will be appropriately addressed in the subsequent identification of assessment endpoints. Consistent with current guidance (U.S. EPA 1992b, 1998a), the additional criteria applied for the selection of site-specific assessment endpoints involves the susceptibility to known or potential stressors.

For purposes of the SLRA, it is appropriate and sufficient to identify the most sensitive endpoints as representative of the ecological values to be protected and to evaluate exposure to the suite of potential contaminants and physical stressors of concern. By screening against appropriately conservative criteria, the analysis will identify and prioritise those stressors that contribute to unacceptable risk, thus providing focus to subsequent phases of the risk assessment.

2.5.3.1 Human Populations

Significant risks to human health are generally considered unacceptable regardless of the location or site-specific conditions. Accordingly, the assessment endpoints for human health are relatively generic, but take into account potential site-specific stressors:

1) Protection of human health from deleterious effects that may result from ingestion or direct contact with environmental media affected by mining operations; and

2) Protection of terrestrial and aquatic resources utilised by local peoples in the middle Fly River and all areas downstream.

2.5.3.2 Terrestrial Ecosystem

Based on the general ecosystem values identified in Section 2.5.2, the following assessment endpoints were identified for the screening level risk assessment:

1) <u>Biological Diversity</u>: Maintain existing species of plants and animals in all functional components of the terrestrial ecosystem;

2) <u>Functional Integrity</u>: Maintain the functional integrity of the terrestrial ecosystem;

3) <u>Nutrient and Energy Cycling</u>: Maintain normal rates of nutrient and energy cycling in terrestrial ecosystems and between terrestrial and aquatic ecosystems; and

4) Sustainable Human Uses of the Terrestrial Ecosystem: Maintain human uses of the terrestrial ecosystem including the gathering of plant products, hunting, and collection of plants and animals for medicinal or ceremonial/spiritual purposes.

These values are comprehensive and embrace both the ecological and human-use-based values of the terrestrial ecosystem. Vegetation structure and diversity of vegetation support diverse animal species, including those used by humans or valued because of their rarity. Biodiversity values are particularly important considerations for assessments involving tropical rain forests (Terborgh 1992; Terborgh and van Schaik 1997; Reagan and Silva del Poso 1995). Protection of plant and animal populations and preservation of rare and threatened species are implicit in biodiversity. Where potential specific effects on particular endpoints at lower levels of ecological organisation are identified in the SLRA, they will be carried forward into the DLRA.

2.5.3.3 Aquatic Ecosystem

Based on the general ecosystem values identified above as well as human use values, assessment endpoints were developed for the aquatic ecosystem:

1) Biological Diversity: Maintain existing species of fish, prawns, zooplankton, insects, phytoplankton, molluscs, reptiles, and amphibians;

2) Functional Integrity: Prevent changes in functional group composition to the extent that these changes negatively impact the composition of another group within the food web;

3) Nutrient and Energy Cycling: Maintain normal rates of nutrient and energy cycling in aquatic ecosystems and between terrestrial and aquatic ecosystems; and

4) Sustainable Artisanal Fishery: Maintain a fish stock that supports small community-based commercial fisheries and maintains the current catch per unit of effort for the artisanal fishery.

2.5.4 Measurement Endpoints

This section describes the endpoints that will actually be measured to ensure that the assessment endpoints are met. A measurement endpoint is a term that refers to a measurable characteristic clearly related to the selected assessment endpoint. Often, assessment endpoints cannot be evaluated directly, and measurement endpoints provide a surrogate.

Measurement endpoints are developed on the basis of relevance to the assessment endpoints, sensitivity, practicality, and other considerations (U.S. EPA 1992a). Measurements for the SLRA should be sufficiently conservative in order to ensure that stressors, pathways, receptors, and areas of potential effect are not inappropriately eliminated from further evaluation.

The remainder of this section describes measurement endpoints for human health, terrestrial vegetation, wildlife, and aquatic life.

2.5.4.1 Human Health

To assess health risks, a chemical's toxicological effects are expressed by the lowest dose eliciting the most sensitive, adverse response (i.e., effect) on people. The most sensitive biological effects are termed measurement endpoints[1]; these are measurable surrogates for the assessment endpoints, as the assessment endpoints cannot directly be measured. Measurement endpoints for human health protect individuals against morbidity. Therefore, they contrast with ecological endpoints, which protect populations against adverse effects to growth, survival, and reproduction.

Toxicity tests are used to define the effects elicited by each chemical; the morbidity or adverse effects (i.e., measurement endpoints) occurring at the lowest doses tested are shown in Section 4 (Table 4). For this SLRA, toxicity values developed by the World Health Organisation or U.S. EPA were selected to quantify these measurement endpoints (Table 13).

The toxicity values represent the amount that can be ingested safely on a daily basis for a long period without adverse effects. The potential for adverse effects from short-term (acute) exposure will not be directly evaluated because acute effects only occur at dosages much higher than those causing long-term (chronic) effects. Thus, chemicals without predicted chronic effects are not expected to cause acute effects (i.e., the chronic toxicity values are also protective against acute effects).

2.5.4.2 Terrestrial Vegetation

Aggradation is the primary physical stressor on terrestrial ecosystems, while flooding and scouring are secondary physical stressors. Therefore, measurements of these factors are appropriate measures of exposure for the SLRA. Evidence of sedimentation and scouring from aerial photography, existing OTML reports (Rau 1995, OTML 1996), and ongoing investigations will be reviewed to determine and document physical stressors. Existing data on sedimentation and flooding, and the resulting models developed by Klohn-Crippen (1996), will be used as benchmarks to evaluate current and future potential for ecological risk for each reach of the Ok Tedi/Fly River systems. Factors such as sediment depth and percentage of time flooded will be compared to the normal range of these factors expected for each river reach. While gardens may be adversely affected by flooding and aggradation, these effects are not strictly ecological and will be addressed in the Social Risk Assessment.

The natural vegetation of the study area is diverse but is poorly known. Information on phytotoxic effects for plant species within the study area is inadequate, but there is a substantial body of literature for contaminant effects on the general types of vegetation present (e.g., monocots, dicots). Therefore, measurement endpoints in the SLRA's risk characterisation for chemical stressors will involve comparisons of the concentrations of chemical stressors in exposure media (soil) within the study area (measures of exposure) to effects threshold concentrations (measures of effect) found in the general literature for toxic effects to vegetation. Because the exposure mechanism is direct

[1] An "endpoint" refers to a specific type of toxic effect on an organ, tissue, or other life process (e.g., survival, reproduction).

contact and the most sensitive indicators will be evaluated, specific plant receptors need not be identified. Evaluation of plant sensitivities to chemical stressors will include both wild and domesticated plant species, thus, species important to humans will be addressed as part of the HERA.

2.5.4.3 Wildlife

Up until this point, the system has been divided into human health, terrestrial and aquatic components. Beginning with development of measurement endpoints and for most of the risk assessment process, an additional distinction will be made. All fauna except for fish and aquatic invertebrates will be collectively called wildlife (e.g., mammals, birds, and reptiles). This organisational change is made because risk assessment methodologies for wildlife, regardless of the habitat in which they occur (e.g., bats vs. crocodiles), are fundamentally the same (i.e., stressor doses are compared to chronic toxicity doses). It therefore makes sense to group these receptors together. During the risk characterisation phase (Section 5), these receptors will be split back out into their respective ecological niches in order to interpret how potential risks to them may influence the overall ecosystem.

Identification of Wildlife Receptors

Receptors are selected because they are either representative wildlife or they are species that are in need of special protection. Valuable species selected here are those used for consumption by native people and/or have other significant uses (e.g., feathers used from great egret [*Ardea alba*]). Species are also chosen because they are important in the food chain, sensitive representatives of entire trophic groups or because they may be exposed to metals from tailings in the water, sediment, or riparian soils that have been inundated by river water.

Wildlife receptors identified for the Fly River include mammalian, avian and reptilian species. Receptor species were identified based on a review of site-specific and general sources of information on wildlife species.

Mammals

Mammals that are important food items, and may have other significant uses were selected as potential receptors. Potential mammalian receptors include wild pigs (*Sus* spp.), rusa deer (*Cervus timoriensis*), and fruit bats (*Dobsonia magna*). Rusa deer and wild pigs were selected as representative mammalian receptors because they can be exposed to metals via drinking water and because they are important food items for local peoples. The fruit bat was selected as a conservative receptor because it is expected to have higher doses than most other mammals (i.e., it has a high ingestion rate-to-body weight ratio).

Birds

Birds that are important food items, have other significant uses (e.g., used for feathers), or have significant potential exposure to mine waste were selected as potential receptors. These avian receptors may forage in the Ok Tedi/Fly Rivers and the Fly River Estuary or use the water as a drinking water source. The great egret was selected as a representative avian receptor because it can be exposed to metals by consuming fish and drinking water. The southern cassowary also was selected as a representative avian receptor because it can be exposed to metals by drinking water from the rivers or their tributaries and they are important food items for humans. Finally, the white-headed stilt (*Himantopus leucocephalus*) was selected as a receptor because it is a probing shorebird that will ingest sediment (potentially high in metals) as well as insects as part of its foraging strategy.

Reptiles and Amphibians

Potential reptilian receptors are divided into three groups that represent complete use of water-dependent food resources in the Fly River -- piscivores, scavengers and herbivores. The New Guinea freshwater crocodile (*Crocodylus novaeguineae*) was selected as a representative fish-eating reptile because it may be exposed to metals by feeding on fish from the river and it is an important food item for humans (Montague 1984). The New Guinea snapping turtle (*Elseya novaeguineae*) was selected as a representative scavenger that also may be exposed to chemicals by feeding on live and dead fish in the lake or river. The herbivorous pig-nosed turtle (*Carettochelys insculpta*) was chosen as a reptilian receptor because it may forage on plants exposed to metals in the river, it inhabits the Fly River (Ernst 1989; Cogger 1994), and it may be used as a food item by humans.

Measurement endpoints for wildlife are focused on model receptors that are important from a human use perspective and/or they are representative of key functional groups identified in the GAE process. The measurement endpoints for these receptors are presented in Table 5.

Table 5. Wildlife measurement endpoints.

Site-Specific Receptor Group	Measurement Endpoint
Herbivorous mammal	Compare stressor dose[1] to a chronic toxicological effect level for herbivorous mammals.
Insectivorous mammal	Compare stressor dose to a chronic toxicological effect level for insectivorous mammals.
Omnivorous mammal	Compare stressor dose to a chronic toxicological effect level for omnivorous mammals.
Fish-eating bird	Compare stressor dose to a chronic toxicological effect level for fish-eating birds.
Herbivorous bird	Compare stressor dose to a chronic toxicological effect level for herbivorous birds.
Probing insectivorous shorebird	Compare stressor dose to a chronic toxicological effect level for insectivorous birds.
Fish-eating reptile	Compare stressor dose to a chronic toxicological effect level for fish-eating reptile
Herbivorous reptile	Compare stressor dose to a chronic toxicological effect level for herbivorous reptiles.
Scavenging reptile	Compare stressor dose to a chronic toxicological effect level for scavenging reptiles.

[1] Dose units are mg/kg-bw/day.

2.5.4.4 Aquatic Life

A very specific series of measurement endpoints have been developed for aquatic life;

1) Potential risks to aquatic biodiversity, ecosystem function, and sustainable fisheries from labile metals will be assessed by comparing metal concentrations with environmental standards that serve as surrogates for protecting the aquatic community;

2) Potential risks from sediment/particulate-associated metals, milling reagents, and total suspended solids to aquatic biodiversity, ecosystem function, and sustainable fisheries will be assessed by comparing concentrations with toxicity thresholds derived from the literature and site-specific data;

3) Potential risks to aquatic biodiversity, ecosystem function, and sustainable fisheries from habitat loss (filling of backwaters and tie channels) will be assessed based on presence/absence of greater than one meter[2] of riverbed aggradation; and

4) Potential risks to aquatic biodiversity, ecosystem function, and sustainable fisheries from vegetation dieback will be assessed in the DLRA.

The first two measurement endpoints involve use of environmental standards. These standards are designed to protect the aquatic community as a whole as well as sensitive and ecologically/economically important species in particular. Consequently, the standards relate directly to the SLRA AEs of biological diversity, functional integrity, nutrient and energy cycling, and a sustainable artisanal fishery.

2.6 CONCEPTUAL MODELS OF HOW MINE-RELATED STRESSORS AFFECT RESOURCES

As part of a risk assessment, graphical models are often developed to illustrate how the stressors are released to the environment and the processes by which people, terrestrial life, and aquatic life are potentially exposed to them. Conceptual models have been developed for human health, aquatic life, wildlife, and terrestrial vegetation. In this section the conceptual models are explained.

Because the stressors and exposure pathways for chemical and physical stressors are quite different, they are treated separately below. However, it is important to recognise that many assessment endpoints may be affected by both physical and chemical stressors.

2.6.1 Human Health

As Figure 7 shows, mine wastes occur in the surface waters and sediments. Due to resuspension and deposition, which depend on streamflows, there is some interchange between these media. People, aquatic life, and wildlife are variably exposed to wastes via the surface water, sediments or by eating aquatic organisms. People will only incidentally ingest constituents in surface water and sediment, but will consume fish and invertebrates harvested from the river. They will be exposed dermally only to surface waters and sediment.

---

[2] Use of one meter of aggradation relative to pre-mine conditions is an admittedly arbitrary threshold selected in the absence of any site-specific data on the effects of aggradation on aquatic habitat.

Many of the villages along the Fly River are expected to have rainwater as a primary source of drinking water (Flew 1998). However, under certain circumstances (i.e., drought) or at villages without rainwater tanks, the use of groundwater from shallow wells as a drinking water source may occur. Therefore, the use of groundwater is a potentially complete exposure pathway assuming that metals in mine tailings may migrate to a well used for drinking water purposes. The potential for exposure by this route is dependent on the specific metal, location of the well, transport of dissolved metals from surface waters bearing mine wastes to groundwater and sorption of metals to soils. Due to these factors, and because groundwater is usually only a secondary source of drinking water, the magnitude of exposures by the groundwater route are not expected to be significant.

The conceptual models for people inhabiting the middle and lower Fly River and the Fly River Estuary are the same. The models assume the following:

- People will be exposed to surface waters, resulting in dermal exposures and incidental ingestion of an occasional mouthful of water; and
- Sediment is abundant, and people will be exposed both dermally and via incidental ingestion while walking, wading, or washing.

2.6.2 Terrestrial Vegetation

The exposure scenario for chemical stressors involves the identification of sources, transport mechanisms, partitioning among environmental media, and exposure pathways to ecological receptors (U.S. EPA 1992b). Conceptual models for chemicals of potential concern are presented in Figure 8.

Physical stressors from mine waste discharges are the solids that enter the stream channels below the mine and are carried downstream in the form of suspended solids and larger particles (Figure 9). The size of suspended particles is related to the velocity of the stream; thus, there is a dynamic equilibrium between suspended and settled materials, depending on the volume and velocity of the water. Suspended solids can produce direct adverse effects on aquatic plants and animals by increasing turbidity that reduces light penetration and obscures visibility.

Aggradation of mine waste occurs in the channel and on the floodplain where water velocity is decreased. This deposition causes direct stress on terrestrial vegetation by burying the ground layer and the roots of trees, woody shrubs, and lianas which results in vegetation dieback. Root zone waterlogging results in anoxic conditions with associated root death and defoliation of trees and lianas (OTML 1996), which in turn results in increased light penetration at ground level, raising the surface temperature which causes rapid drying of the sediment layer.

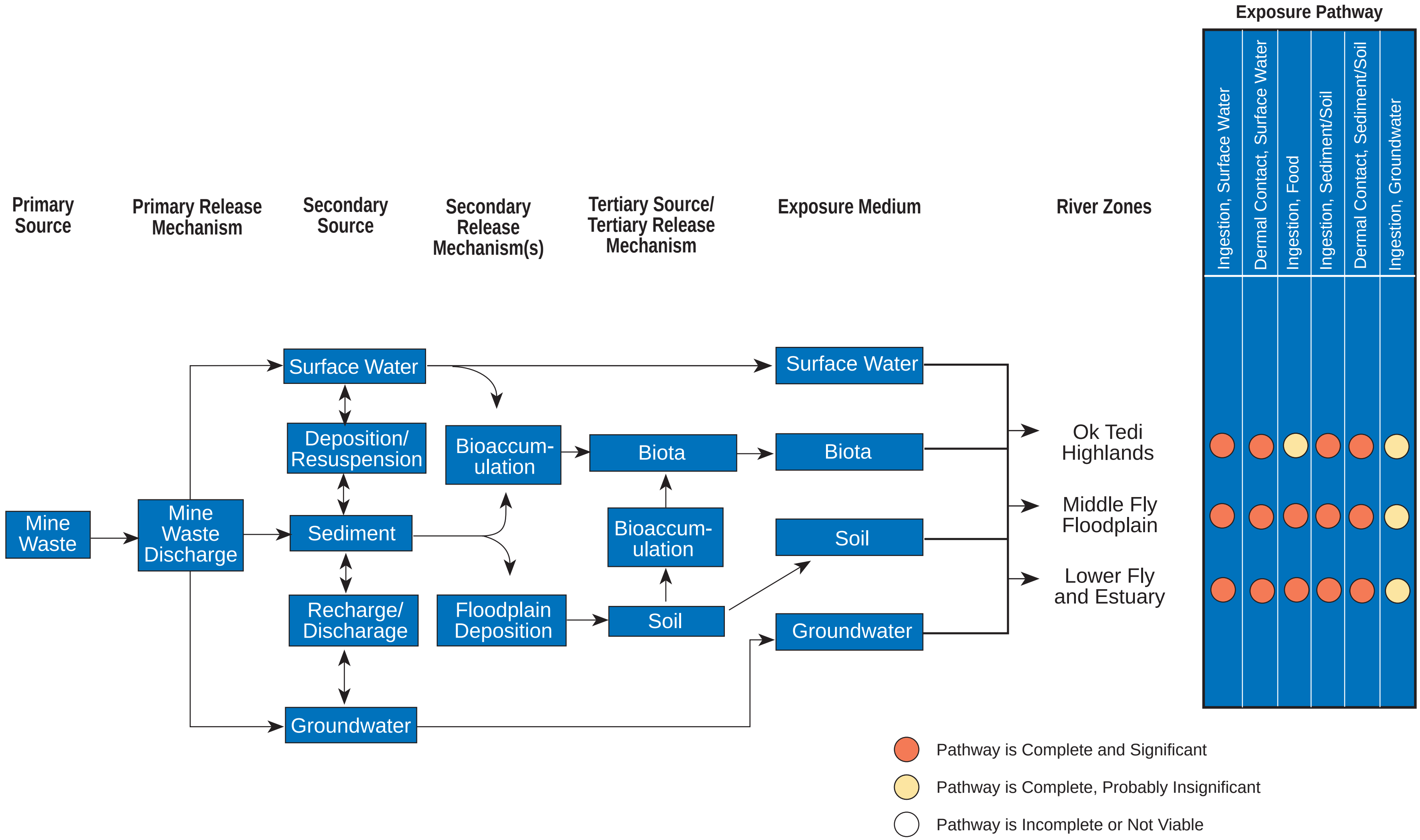


Figure 7.
Human Health Conceptual Model for Chemical Stressors

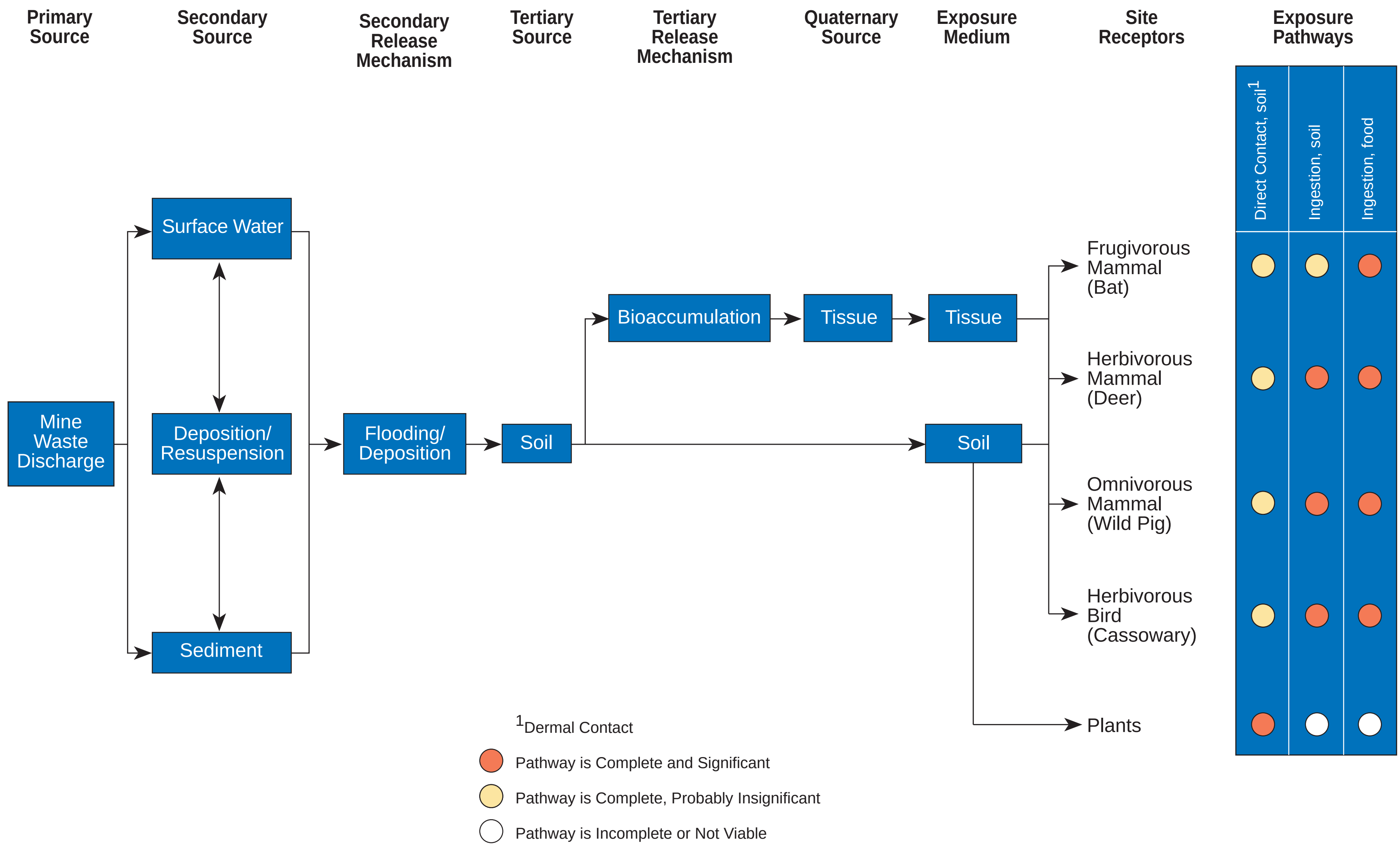
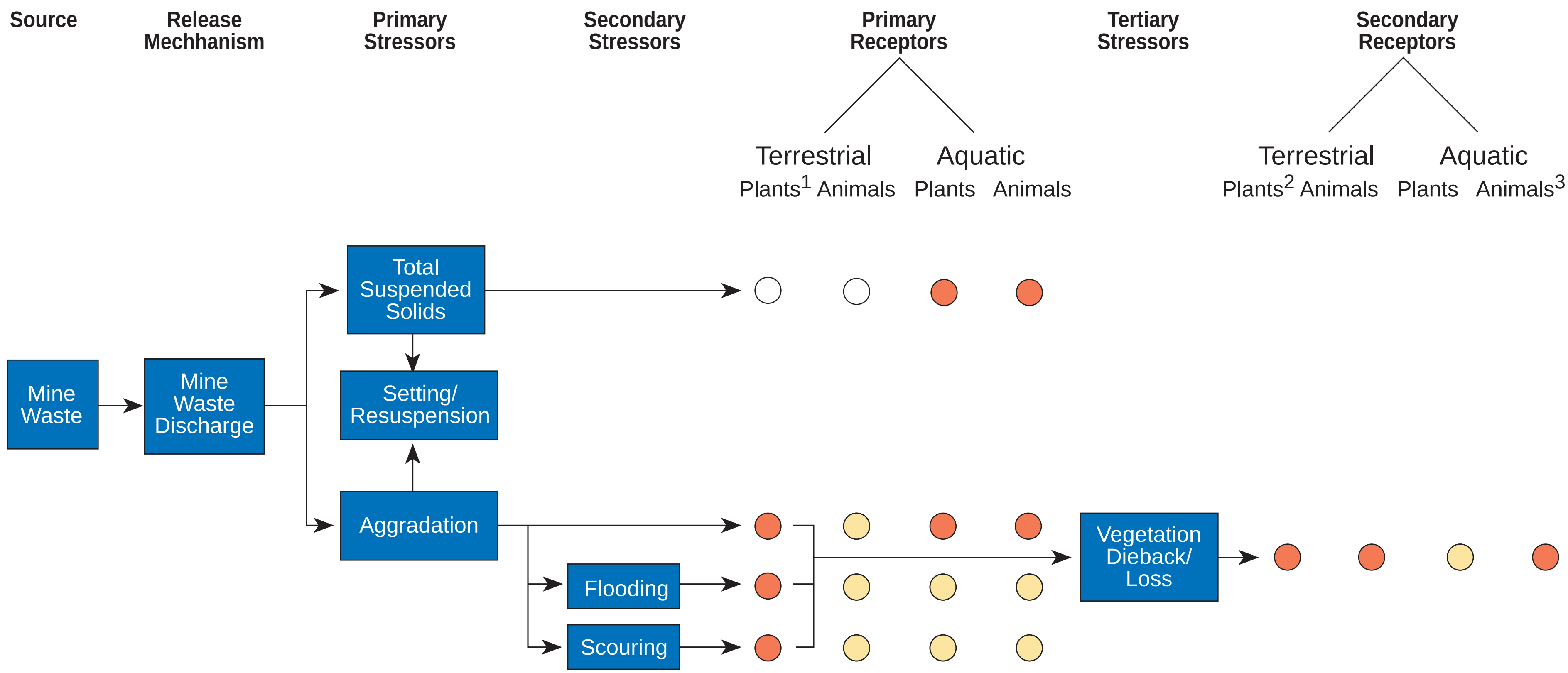


Figure 8.
Terrestrial Ecosystem Conceptual Model for Chemical Stressors



Pathway is Complete and Significant

Pathway is Complete, Probably Insignificant

Pathway is Incomplete or Not Viable

[1] Ground Layer, Woody Shrubs, Trees, Lianas

[2] Ground Layer, Lianas, Epiphytes

[3] Primary Fish that Forage on Terrestrial Fruits and Insects in the Floodplain

**Figure 9.
Terrestrial and Aquatic Ecosystem Conceptual Model for Physical Stressors**

Aggradation of the channels in some reaches raises the water level, causing increased flooding. The increased areas and percent time of flooding stresses the vegetation that are not adapted to prolonged inundation, causes dieback. Adverse effects of flooding compound the effects of aggradation. Water flowing down tributary streams is backed up at the mouth of streams entering portions of the floodplain as a result of main channel flooding, thereby flooding the tributaries and causing dieback in areas where mine waste aggradation has not occurred.

Raised water levels increase the flows along channel banks and across the narrow necks of land created by meanders in the floodplain of the lower Ok Tedi and upper middle Fly River. This scouring accelerates cutbank formation, uproots trees, exposes roots, and destroys seedlings and saplings.

Vegetation dieback and loss of vegetation along channels becomes a tertiary stressor to the terrestrial ecosystems by reducing cover and limiting the production of flowers, seeds, and fruits. These changes adversely affect epiphytic plants, insects, bats, tree kangaroos, cuscus, birds, and other animals that depend on the floodplain forest for food and shelter. Fish species that feed on fallen fruit and eat terrestrial insects in flooded forest habitat are also adversely affected by vegetation dieback.

Dead forest vegetation is replaced by more flood-tolerant grassland savannah vegetation, and grassland vegetation is replaced by aquatic vegetation in areas that become permanently flooded to a sufficient depth. These conversions result in changes in both the plant and animal resources available to humans such as game for food, plants for food, and plant and animal products used for medicinal or ceremonial purposes. These changes can result in shifts in food types, accessibility to needed resources, and/or the loss of these resources within clan territorial boundaries.

2.6.3 Aquatic Life

The aquatic life conceptual model for chemical stressors is shown in Figure 10. As shown, chemicals leaching from mine tailings may dissolve in surface water or pore water or be adsorbed to suspended solids or settled solids (sediment). Aquatic life are mostly exposed to dissolved chemicals via the gill (pelagic and epibenthic organisms exposed to surface water, and infaunal benthos exposed to surface water, pore water, or a combination of the two). Chemicals adsorbed to solids may also be taken up by organisms via ingestion. The model also shows that metals incorporated and retained by prey organisms represents another exposure pathway for aquatic life (i.e., through food ingestion).

Non-chemical stressors (e.g., total suspended solids, sedimentation) may affect aquatic life directly or indirectly (see Figure 9). Direct effects include smothering and suffocation, while vegetation dieback (biological stressor) is an indirect effect that results in a loss of aquatic life habitat for reproduction and feeding.

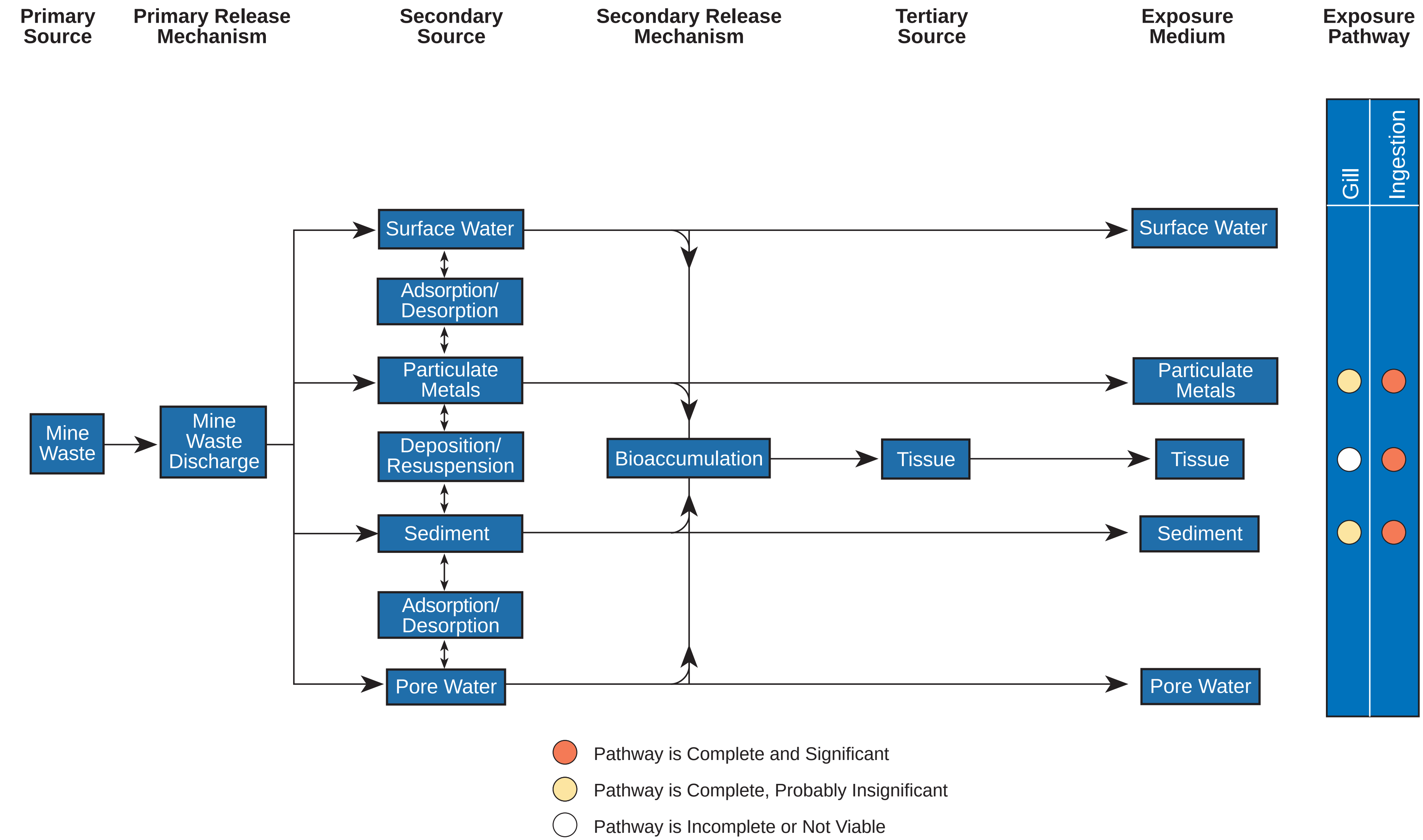


Figure 10.
Aquatic Life (Fish and Invertebrates) Conceptual Model for Chemical Stressors

2.6.4 Wildlife

Wildlife may be exposed to stressors in the aquatic habitat through direct ingestion of aquatic food items and ingestion of water (Figure 11). Reptiles and birds may be exposed in the aquatic habitat. Furthermore, terrestrial wildlife may be exposed through direct ingestion of water and food items, as shown in the terrestrial conceptual site model (see Figure 8). These pathways were chosen since they represent receptor groups that can feed from these habitats (habitats affected by tailings) and can be exposed to chemicals from the tailings.

The significance of each exposure pathway was considered relative to other pathways for a given receptor. For example, exposure to metals via the food for wildlife is considered much more important than that via direct contact.

2.6.5 Stressor-related Interactions Between Receptor Groups

A final consideration in the conceptual model process is how potential direct effects to one receptor group (e.g., terrestrial vegetation, aquatic life) may indirectly affect another receptor group. The pathways for direct effects were described in the previous models. Figures 12 and 13 provide a conceptualisation of indirect effects. Most of these indirect effects are intuitively obvious, but it is important for them to be explicitly addressed during the risk characterisation phase of the assessment. All of the indirect effects are a result of either loss of habitat or food resources.

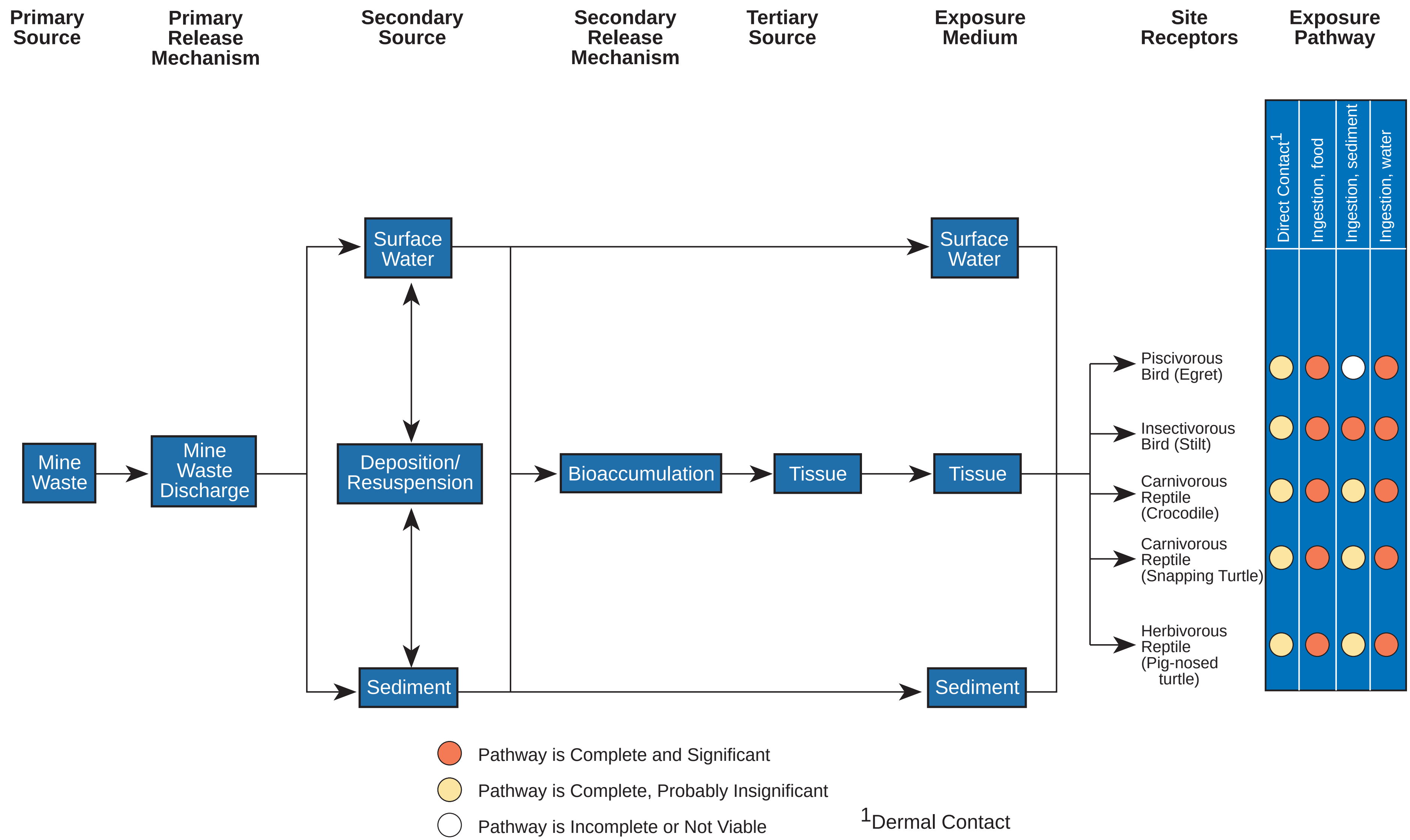


Figure 11. Aquatic Ecosystem Conceptual Site Model for Wildlife

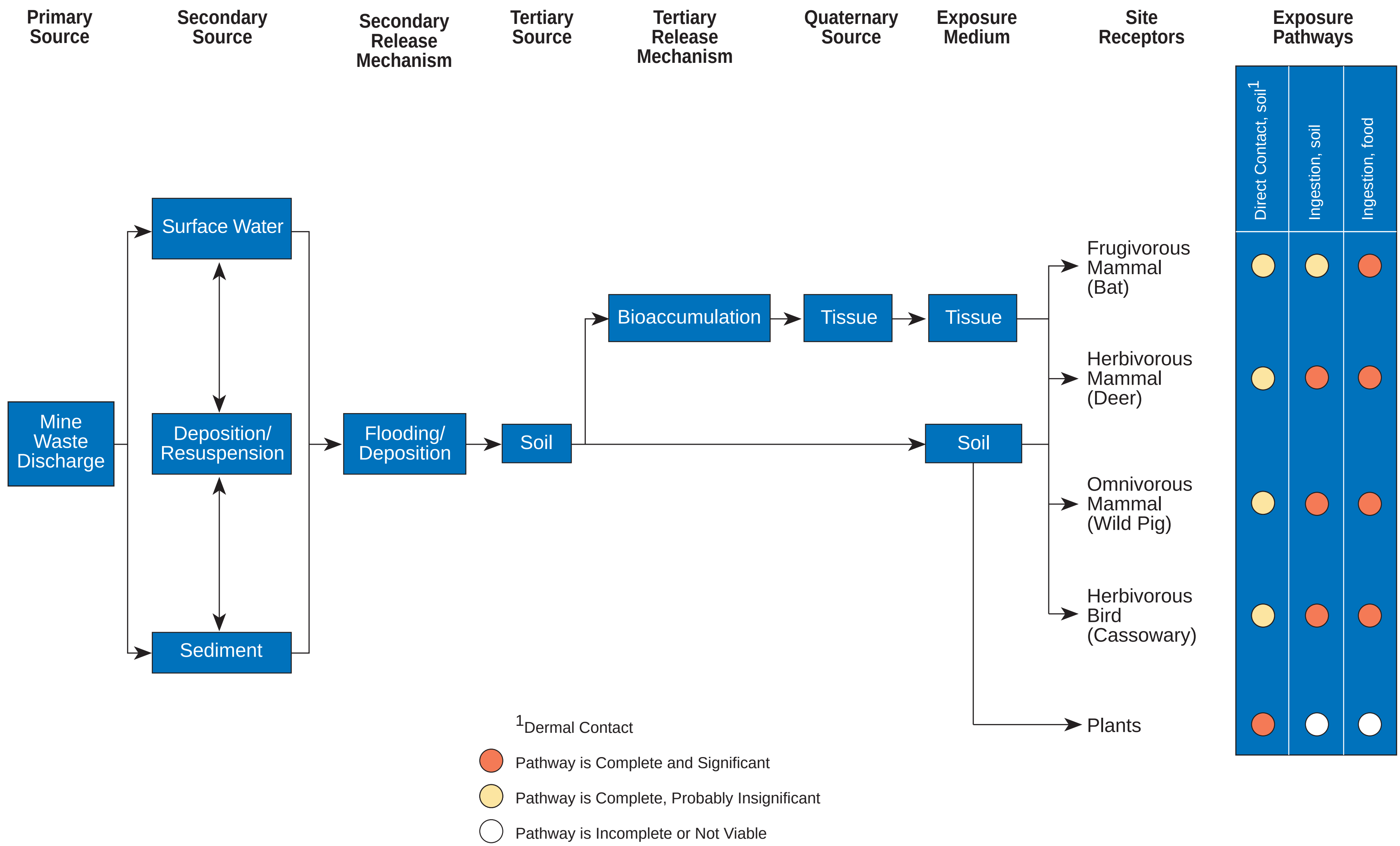
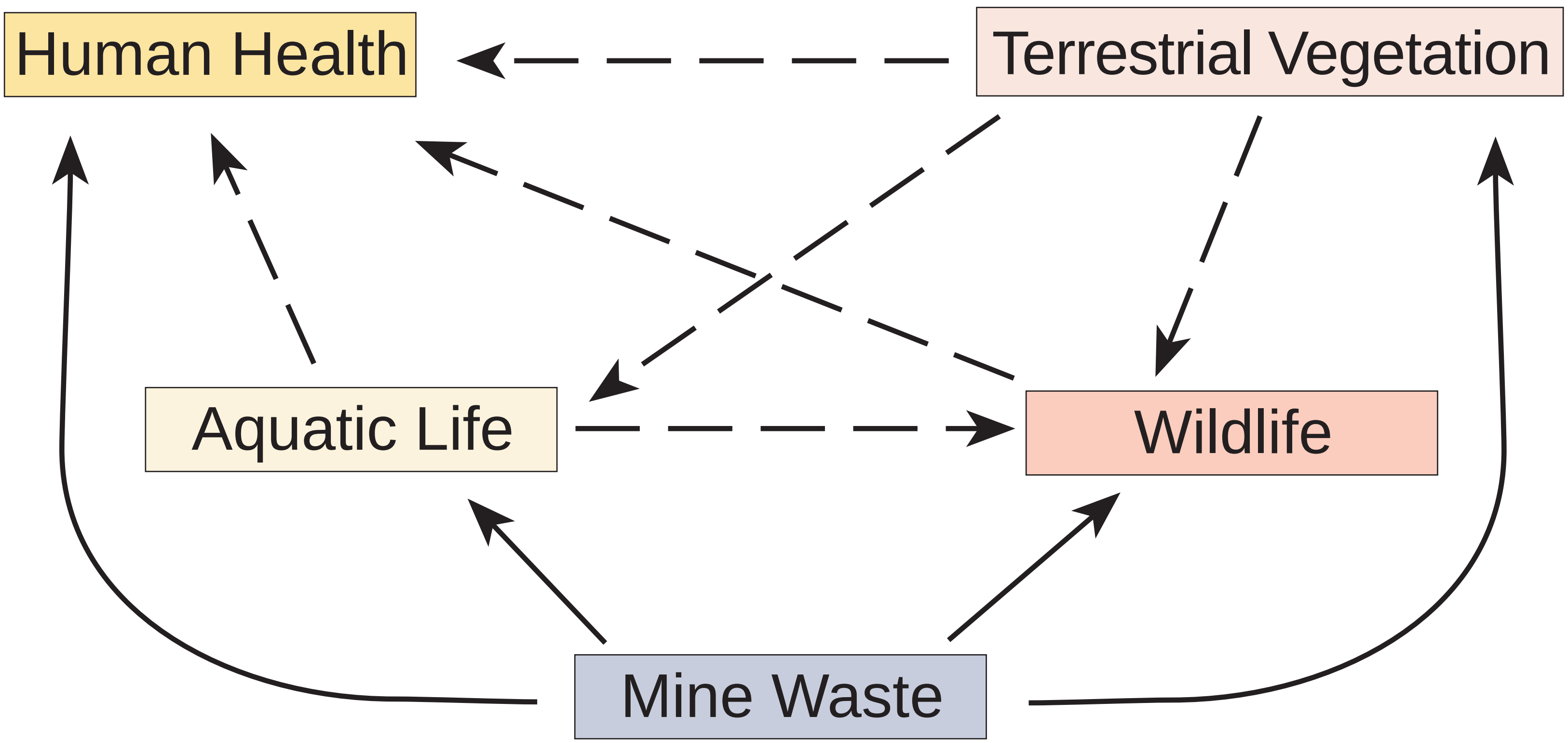


Figure 12. Terrestrial Ecosystem Conceptual Model for Chemical Stressors



Direct Effect

Indirect Effect

Figure 13.
Conceptual Model of Direct and Indirect Effects on Human Health and Ecological Receptors

3. EXPOSURE CHARACTERISATION

The purpose of the Exposure Characterisation is to describe (normally in a quantitative manner) the degree or extent to which receptors are exposed to the various SOPCs. Different approaches are used for the different receptor groups such as people, terrestrial vegetation, aquatic life and wildlife as described in the remainder of this section. As a general approach to screen the various chemical and physical stressors in the river system, the Ok Tedi/Fly River system was divided into six river reaches, plus the estuary. These reaches were defined based on various physical and biological differences that are observed at different sections of the river (Figure 14).

3.1 DATA SOURCES

A variety of data sources were used in the Exposure Characterisation phase of the SLRA. Analytical data was taken from OTML reports (OTML 1997b; 1998c), as well as electronically transmitted data from OTML in 1998 and 1999, and CSIRO in 1999. These data represent chemical concentrations in sediments, surface waters, fish and shellfish tissue, and cassava and sago. In addition a small amount of drinking water data was taken from Flew (1998) and sediment data was collected from a United Nations report (UNEP 1995).

Collectively, these data were used to estimate concentrations to which people and the various ecological receptors could be exposed. The OTML sediment, tissue and surface water data were collected for purposes other than risk assessment, and while these data may be used for assessing risk, they do not completely cover all media, river reaches, and chemicals identified for assessment in the Problem Formulation phase (see appendices). Thus, there are some significant data gaps which prevent a full evaluation of potential risks.

3.2 HUMAN HEALTH

This section explains methods used to estimate the amount of a chemical stressor people could take into their bodies from contacting water, standing or working in sediment and water, and eating foods potentially exposed to mine waste. This estimate is referred to as an intake rate. The following section describes, both qualitatively and quantitatively, the exposure pathways by which these populations are potentially exposed.

3.2.1 Potentially Exposed Human Populations

People living in villages near mine impacted environmental media (e.g., rivers carrying tailings) are potentially exposed to chemical stressors. The potentially exposed populations for this risk assessment have been defined as those people living along or nearby the Ok Tedi and Fly Rivers as well as the estuary. Because uses of the rivers and the mine affected media, and the intensity of uses differ along the various reaches of the river system, people living along each of the river reaches have been considered a separate exposed population. Although it is possible that people in other areas could come into contact with mine impacted media, it is expected that the

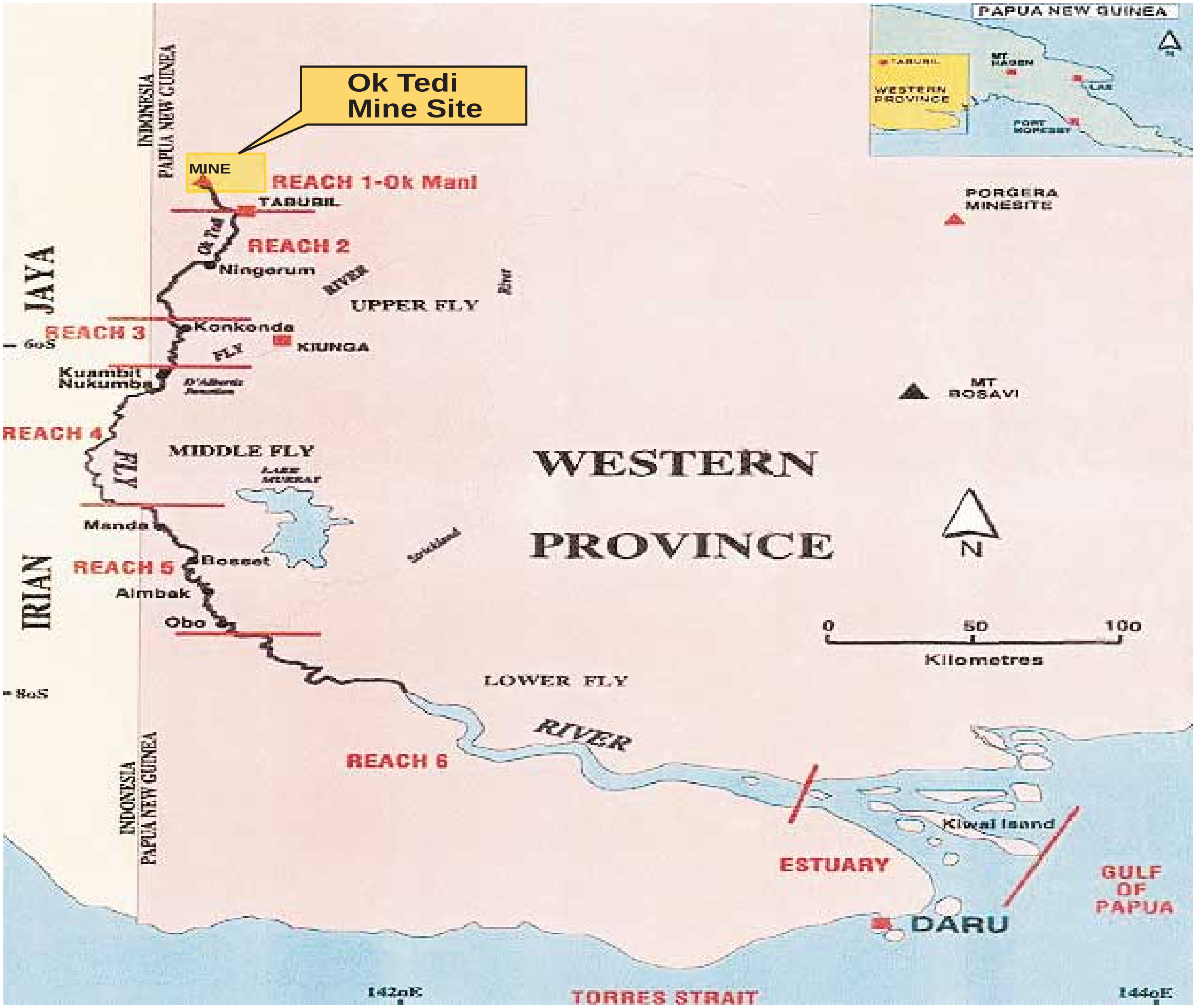


Figure 14.
River Reaches Evaluated
in the Ok Tedi/Fly River System

defined "receptor" populations would have the highest intensity of river use and will have correspondingly higher exposures and risks. Based on the foregoing, any exposed individuals or populations not characterised in the risk assessment are expected to have lower risks than those presented here.

Exposures to all segments of the human populations have been considered, including those having highest exposure or who are especially sensitive. Accordingly, children appear to be a sensitive group. Exposures (e.g., metal intakes) of children may be greater than those occurring in adults, due to their lower body weights, larger body surface area in relation to weight and the tendency for chemical stressors to be absorbed more readily by children than by adults (WHO 1986; Bearer, 1995). As a result of these factors, children have higher intakes on a per unit of body mass basis (i.e., mg/kg-bw/day). Additionally, child behaviours can result in increased contact rates with affected environmental media (e.g., water and sediment) that would in some cases increase their exposure over adults. For the purposes of this risk assessment, both of these factors have been taken into consideration. For these reasons, exposures have been calculated as a weighted average accounting for exposure occurring during both childhood and adulthood. Additionally, to ensure that no potential risks to children have been overlooked, exposures occurring during childhood have been estimated separately (Section 5.5.5.1).

The potentially exposed populations were separated into three groups based on (1) geographic location along the River, and (2) expected differences in body weights and food consumption rates. These three River areas are described in detail in Problem Formulation and are summarised below:

1) People living along the Ok Tedi River from Tabubil to the D'Albertis junction;

2) People living along the middle and lower Fly River from D'Albertis junction out to the Fly River Estuary; and

3) People living along the Fly River Estuary.

3.2.2 Types of Exposures Evaluated

The discussion that follows focuses in greater detail on the potentially tailings-impacted media and exposure pathways. Issues applicable to all exposures are discussed first, then media-specific exposure issues. People may contact tailings in a variety of ways. The high suspended sediment load in the OK Tedi and Fly Rivers makes them unappealing for drinking and clothes washing, but other activities such as fishing, wood gathering, and hunting may occur with greater frequency. Exposure to chemicals in tailings is most likely to be associated with capture and consumption of fish, molluscs, and shellfish as well as incidental contact with water and sediment while engaging in activities near the impacted rivers (e.g., fishing or playing). Exposures that are related to tailings-affected media, and where contact may occur are referred to as complete exposure pathways while exposures that are not related to tailings or cannot occur because there is no way for people to be exposed are referred to as incomplete.

As described in Section 2.4, tailings flow from the source at the mine to the Fly River and estuary. People may have skin contact with surface water while bathing, swimming, or performing other activities, and they are expected to occasionally accidentally ingest a mouthful of water while engaging in these activities. Therefore, the skin and incidental water ingestion pathways were considered complete. The source of drinking water for villages is not well characterised. However, both surface water and rainwater tanks may be used (Flew 1998). Due to the possibility that surface waters maybe used, this pathway is potentially complete. However, drinking water data is limited to a few samples of copper and lead taken in 1998 (Flew 1998).

Local people may engage in various activities that may bring them into contact with mine-related sediments or soils through skin contact and incidental ingestion. These activities may include wading along riverbanks or gardening. Therefore, sediment exposure pathways are complete.

Fish and shellfish may accumulate mine-related metals from the water, sediment, or food (their prey). The consumption of fish and other aquatic foods is thus a complete pathway. Gardening may occur on soil containing tailings at some villages. Due to the possibility of tailings deposition, exposure to tailings-related chemicals through consumption of food grown on tailings (e.g., sago, cassava) is considered a complete pathway.

3.2.3 Exposure Parameters

Human intake of chemical stressors was calculated using equations consistent with international practices for human health risk assessment (European Commission 1996; U.S. EPA 1989b). Chronic daily intake is expressed in terms of milligrams of a chemical stressor absorbed into the body per kilogram of body weight per day (i.e., mg/kg-bw/day). Intakes are calculated for each chemical stressor and complete exposure pathway. Estimating daily intake required a number of specific assumptions, some common to all river reaches (Table 6). Assumptions were based on a combination of generic and, when available, site-specific data.

For the purposes of assessing the aquatic food consumption pathway, it was assumed that both fish and shellfish could constitute part of the diet where both these tissue types were available (i.e., the estuary).

Table 6. Human health exposure assumptions.

Parameter	Parameter Value: Adult	Parameter Value: Child	Reference
Exposure Duration	65 yr	5 yr	BPJ[a]
Averaging Time (non- cancer endpoints)	65 yr	5 yr	BPJ
Averaging Time (cancer endpoints)	70 yr	70 yr	BPJ
Frequency of Exposure (food and water consumption)	260/365 day/yr[b]	260/365 day/yr[b]	BPJ; Jackson 1998
Exposure Time (water ingestion)	4 hr/day	4 hr/day	BPJ
Ingestion Rate (water)	25 ml/hr	25 ml/hr	BPJ
Frequency of Exposure (incidental exposure to sediments or water)	260 day/yr	260 day/yr	BPJ
Sediment Loading (to skin)	0.132 mg/cm^2	12 mg/cm^2	U.S. EPA 1996b; BPJ
Absorbed fraction from sediment	0.005	0.005	U.S. EPA 1992a
Ingestion Rate (sediment)	100 mg/day	200 mg/day	BPJ; U.S. EPA 1996b
Ingestion Rate (drinking water)	4 L/day	2L/day	BPJ
Partial Exposed Surface Area (skin-sediment exposure-Ok Tedi and middle Fly River)	4,834 cm^2	2,788 cm^2	U.S. EPA 1996b; U.S. EPA 1985f; CSIRO 1996; BPJ
Partial Exposed Surface Area (skin-sediment exposure-middle, lower Fly River and Estuary)	4,998 cm^2	2,788 cm^2	U.S. EPA 1996b; U.S. EPA 1985f; CSIRO 1996; BPJ
Full Body Exposed Surface Area(skin) (Ok Tedi and middle Fly River Only-water exposure)	15, 500 cm^2	6,835 cm^2	U.S. EPA 1996b; U.S. EPA 1985f; CSIRO 1996; BPJ
Full Body Exposed Surface Area(skin) (lower Fly River and Estuary-water exposure)	16,050 cm^2	6,835 cm^2	U.S. EPA 1996b; CSIRO 1996; BPJ
Body Weight (Ok Tedi and middle Fly River)	47 kg	10 kg	CSIRO 1996; Flew 1998
Body Weight (lower Fly River and Estuary)	55 kg	10 kg	CSIRO 1996; Flew 1998
Shell Fish Ingestion Rate (Estuary)	140 g/day	23 g/day	Yok 1990b; BPJ
Fish Ingestion Rate (Ok Tedi and middle Fly River)	245 g/day	45 g/day	OTML 1998a; BPJ
Fish Ingestion Rate (Estuary)	126 g/day	25 g/day	Yok 1990b; BPJ
Fish Ingestion Rate (lower Fly River)	282 g/day	51 g/day	Yok 1990b; BPJ
Sago Ingestion Rate (Ok Tedi and Fly River)	235 g/day	145 g/day	BPJ
Cassava Ingestion Rate (Ok Tedi and Fly River)	60 g/day	30 g/day	BPJ

[a] BPJ = Best Professional Judgement
[b] 260 d/yr in Reach 3 only, based on data from Jackson 1998.

3.2.4 Calculation of Stressor Intakes

Exposure pathways evaluated are the same for all exposed populations in the study area. The conceptual exposure models are presented in the Problem Formulation. To address differences in body weight and intake rates that vary by age, the intakes were calculated as weighted averages over each of the estimated durations of childhood and adulthood using Equations 1 through 4 below. Equation 1 was used to estimate ingestion intakes (e.g., sediment, water, or food) averaged over a long-term exposure.

$$Ingestion\,CDI = \frac{EEC\ x\ \ IR\ x\ \ EF\ x\ \ ED\ x\ \ CF}{BW\ \ \ x\ \ \ AT} \qquad (1)$$

where:

CDI	=	Chronic daily intake, mg/kg-bw/day
EEC	=	Expected environmental concentration in medium, mg/kg or mg/L
IR	=	Contact rate, L/day, or g/day for water, or soil and food ingestion, respectively.
EF	=	Exposure frequency, day/yr
ED	=	Exposure duration, years
CF	=	Conversion factor 10^{-3} L
BW	=	Body weight, kg
AT	=	Averaging time, years x 365 day/yr

The methods used to estimate average daily chronic human health intake for exposures involving skin contact with water and sediments are shown in Equations (2) and (3), respectively. Metal uptake across the skin also is known as dermal uptake.

$$Dermal\ CDI_{\ (water)} = \frac{EEC\ x\ SA\ x\ Kp\ x\ ET\ x\ EF\ x\ ED\ x\ CF}{BW\ x\ AT} \qquad (2)$$

where :

CDI	=	Chronic daily intake, mg/kg-bw/day
EEC	=	Expected Environmental Concentration of chemical dissolved in water, mg/L
SA	=	Skin surface area available for contact, cm^2
Kp	=	Chemical-specific permeability coefficient, cm/hr
ET	=	Event time, hr exposure/event
EF	=	Exposure frequency, exposure events/yr
ED	=	Exposure duration, yr
CF	=	Conversion factor, 1 L/1000 cm^3
BW	=	Body weight, kg
AT	=	Averaging time, yr x 365 d/yr

$$Dermal\ CDI_{(sediment)} = \frac{EEC\ x\ SA\ x\ AF\ x\ ABS\ x\ EF\ x\ ED\ x\ CF}{BW\ x\ AT} \qquad (3)$$

where :

CDI	=	Chronic daily intake, mg/kg-bw/day
EEC	=	Expected Environmental Concentration of chemical in sediment, mg/kg
SA	=	Skin surface area available for contact, cm^2
AF	=	Sediment-to-skin loading rate, mg/(cm^2-event)
ABS	=	Chemical-specific absorption factor, unitless
EF	=	Event frequency, events/yr
ED	=	Exposure duration, yr
CF	=	Conversion factor, 1E-06 kg/mg
BW	=	Body weight, kg
AT	=	Averaging time, yr x 365 d/yr

For all exposure pathways, the adult and child intake estimates were weighted to a total lifetime chronic exposure reflecting the relative portion of the expected lifespan that a person may be exposed during both childhood and adulthood. Equation 4 was used to calculate the weighted averages. For the calculation of child-specific dose (Secton 5.5.5.1) no weighting was done and all doses were calculated using equations 1-3 using child specific exposure parameters (Table 6).

$$Lifetime\ Average\ CDI = CDI_{adult}\ x\ (ED_{adult}/lifespan) + CDI_{child}\ x\ (ED_{child}/lifespan) \qquad (4)$$

where:

CDI	=	Chronic daily intake, mg/kg-bw/day
ED	=	Exposure duration, years
Lifespan	=	Lifespan estimate, 70 years

3.2.5 <u>Exposure Concentrations</u>

Concentrations of chemical stressors to which people might be exposed were calculated using the monitoring data from 1994-1996. Exposure concentrations in each medium (e.g., sediment, water, fish) were estimated as the 95 percent upper confidence limit (UCL)[3] of the mean (Equation 5).

---

[3] The 95% UCL of the mean is needed because it in not possible to know the true mean. The 95% UCL of a mean is technically defined as a value that, when calculated repeatedly for randomly drawn subsets of site data, equals or exceeds the true mean 95 percent of the time.

$$95UCL = C + (t \times SE) \qquad (5)$$

Where:

95UCL	=	95 percent upper confidence limit of the mean expected environmental concentration
C	=	Mean concentration
t	=	t value, based on sample size
SE	=	Standard error of expected environmental concentration

When only one sample was available, the observed concentration was used as the exposure concentration for each chemical stressor. In cases where a chemical was undetected, it was assumed for the purposes of the risk assessment that it was present at one-half the analytical detection limit.

A summary of the 95 percent UCL data by exposure media and river reach can be found in Appendix A.

3.2.5.1 Sediment and Tissue Moisture Content

For the human health risk assessment it was necessary to convert dry sediment and tissue concentrations to a wet weight basis. For sediment, no site-specific moisture data were available at the time of this report. Therefore, it was necessary to make a conservative assumption regarding the amount of moisture in the sediment samples. Based on limited sediment moisture data from the nearby lower Strickland River (n=64), it was conservatively assumed that the samples had a 30 percent moisture content.

Limited moisture data were available for tissue. Since these data were not specific to the samples being used in the risk assessment but were generic to the river system, an average moisture content was calculated and used to convert dry to wet weights. The estimated tissue moisture content was approximately 84 percent (Tinkerame 1998, personal communication).

3.2.5.2 Fish Tissue Concentrations

Both fish flesh and fish liver concentrations were available for the SLRA. Total fish tissue concentrations were estimated by assuming that the flesh and liver concentrations represented 99 and 1 percent of the total fish concentrations, respectively (OTML 1998a).

3.3 TERRESTRIAL VEGETATION

As discussed in the Problem Formulation, risk to terrestrial vegetation is potentially associated with both physical and chemical stressors. Because the methods for assessing these potential risks to terrestrial vegetation are fundamentally different, the exposure, effects, and risk characterisation components for terrestrial vegetation are divided between physical and chemical stressors.

3.3.1 Physical Stressors

Potentially affected terrestrial vegetation (receptors) in the HERA study area are defined as the emergent vegetation (e.g., mangroves, monsoonal savannah) or upland vegetation on lands adjacent to the Ok Tedi and Fly River systems that are potentially exposed to mine-related stressors. Adverse effects on vegetation can indirectly affect the animal community of terrestrial ecosystems and even impact aquatic food webs. These exposure pathways are indicated in the conceptual models that were presented in the draft Problem Formulation document.

Aggradation is the primary physical stressor affecting terrestrial vegetation, which in turn creates flooding and scouring as secondary physical stressors. The parameters for defining physical processes as stressors and for quantifying exposure are:

- severity (i.e., intensity);
- spatial extent; and
- temporal duration.

Aggradation, flooding, and scouring are defined as physical stressors of terrestrial vegetation on the basis of increases in these selected parameters of natural hydrologic processes. Parameters used to define physical stressors are described in subsequent paragraphs.

Receptor contact with these physical stressors occurs during periods of high water that inundates areas of terrestrial vegetation, and contact with sediment that continues after floodwaters have receded. Floodplain aggradation deposits sediment directly onto surface vegetation and buries surface litter, soil organisms, and the roots of woody vegetation in floodplain areas adjacent to the river channels. However, flooding extends over broader areas of the floodplain and may back up water into tributary streams. The extended annual duration of mine-induced flooding in low-lying areas produces vegetation stress. Scouring occurs where water levels are elevated to the point that flow rates increase along channel banks and across the necks of meanders as the result of flooding caused by river bed aggradation.

Sediment depth is the appropriate measure of severity for floodplain aggradation. While natural sediment deposition occurs normally on the floodplain and along the shoreline of rivers and related water bodies, mine wastes have added incrementally to the normal sediment accumulation in these areas (Klohn-Crippen 1996). It is the incremental aggradation along the shoreline and on the floodplain (i.e., increased sediment depths in terrestrial ecosystems that are the result of mine waste disposal activities) that can cause direct adverse effects on terrestrial vegetation.

Flooding frequency is defined as the percentage of time, nominally reported on an annual basis, that flood events occur (Marshall and Rau 1999). While water depth may be important, it is the duration of flooding on an annual basis (i.e., flooding frequency) that causes vegetation stress and is, therefore, the appropriate exposure parameter indicative of flooding stress. Terrestrial vegetation bordering steep channels and on floodplains is normally flooded for intervals throughout the year. However, mine-related aggradation in the stream channel and on the floodplain has increased the duration of annual flooding in some areas, causing adverse effects (e.g., vegetation dieback) as described in Section 4.2.2. Stressed terrestrial vegetation in areas not subject to mine-related sediment deposition (e.g., into tributary streams) provide evidence that flooding stress may be more extensive than direct stress from floodplain aggradation.

The increased river height caused by riverbed aggradation increases the rate of water movement across the necks of meanders and along riverbanks. These increased flows cause, or at least exacerbate, scouring effects. While scouring stress is easily identified, quantification of exposure above normal rates of occurrence is problematic. Exposure to scouring stress has been identified on the basis of direct observation in areas where sedimentation and flooding from mine-related activities is occurring.

Identification of these physical stressors is site-specific (i.e., exceedances of normal levels of these natural processes creates the stress). It is therefore necessary to understand the normal levels and range of values for stressor parameters in order both to define the stressors and to evaluate the stresses. Section 4.2.2 describes the process for establishing effects thresholds for physical stressors on terrestrial vegetation.

3.3.2 Chemical Stressors

The potential risks to terrestrial plants posed by chemical SOPCs in sediments in the Ok Tedi/Fly River system have been assessed. As discussed above, the river system was partitioned into six river reaches and the Fly River estuary based on similarity of geographical features and association with tributaries.

Sources of sediment data included the OTML Annual Report 1995/6 (OTML 1996), the United National Environmental Programme in 1995 (UNEP 1995), and samples analysed by CSIRO in 1998. The sediment data used in this assessment were collected primarily from years 1994-1996 and 1998. Sediments collected by UNEP (1995) were collected in the early 1990's using a 36-mm diameter gravity corer with an acrylic liner. Mine-derived sediments in the upper portion of cores were wet-sieved with the <20 µm fraction saved for digestion in aqua regia and chemical analysis. Collection and sample processing techniques used for the OTML (1996) and CSIRO (1998) sediment data are unknown.

Terrestrial vegetation communities occur primarily on the riverbanks, levees, and floodplains of the river system. Consequently, sediment data from these areas was used in preference to other sediment data available (i.e., mid-channel and ORWBs).

Floodplain sediment data were available for the SOPCs in Reaches 4 and 5. No sediment data were available from Reaches 1 (Ok Mani), 3 (lower Ok Tedi), or 6 (lower Fly River). Riverbank sediment data were available for the SOPCs in Reaches 2, 4 (upper middle Fly River), and 5 (lower middle Fly River). Mid-channel sediment data were available for iron in Reach 5, and for copper in estuary sediments. ORWB sediment data were available only for copper in Reach 4.

During flood events, suspended sediments from the river are deposited on the riverbanks and low-lying areas in Reaches 1 and 2, and on the riverbanks, levees, and floodplains down river. The sediment data from Reach 2 represents mid-channel sediments, primarily as this reach does not have appreciable off-river water bodies or floodplain areas and is, in general, not a sediment depositional area (except in the river channel proper). It is likely that only riparian vegetation communities occurring along the riverbanks have been exposed to river sediments.

To estimate potential SOPC exposure concentrations, the 95 percent upper confidence level of the mean (95 percent UCL) of the combined data for each river reach was calculated and used as the estimated concentration to which plants may be exposed. A summary of the estimated SOPC exposure concentrations for the various chemical stressors and river reaches evaluated can be found in Appendix B.

3.4 AQUATIC LIFE

The SLRA Exposure Characterisation for aquatic life included quantifying exposure to chemical SOPCs in surface waters and sediments, as well as exposure to TSS, low dissolved oxygen (DO) levels, and habitat loss via river bed aggradation. Two SOPCs – vegetation dieback and pH were not assessed. Vegetation dieback is inherently screened into the DLRA but cannot be quantitatively evaluated from an aquatic life perspective until the DLRA. Similarly, potential risk from pH cannot be evaluated until completion of ongoing studies evaluating the potential for ARD.

Based on visual inspection of temporal and spatial data plots, chemical concentrations in water and sediment did not exhibit significant within reach relationships. Consequently, it appeared appropriate to use all data within a reach collected from 1994-1996 to characterise exposure (as well as the sediment data for samples collected in 1998 by CSIRO). The 95 percent UCL of the mean was used to estimate chronic exposure concentrations in surface water and sediment, while the 95th percentile of all the data for a reach was used to estimate potential acute exposures (surface water only). Acute exposures in sediment were not evaluated because chemical concentrations in sediment are a result of long-term deposition. Accordingly, benthos are not likely to have acute exposures that are greater than chronic exposures. Unlike the chemical data, the TSS data did exhibit a significant within reach spatial relationship. Consequently, TSS exposure was characterised on a station-by-station basis rather than by reach.

3.5 WILDLIFE

3.5.1 River Reaches

Like the other SLRA components, the wildlife risk assessment divided the Ok Tedi and Fly River system into separate reaches. Each model receptor is expected to inhabit a particular area based on the food they consume and habitat requirements. For example, the freshwater crocodile is expected to inhabit both the river main stem as well as ORWBs. The river reaches each receptor inhabits are shown in Table 7.

3.5.2 Dose Estimates for Wildlife

Metals measured in surface waters, tissue, and sediment were evaluated in the wildlife risk assessment. However, not all wildlife receptors were assumed to be exposed to all three media; and different receptors were exposed to different food sources. For example, the white-headed stilt is the only receptor expected to consume invertebrates and to incidentally ingest sediment. As for terrestrial vegetation, it was assumed that soil invertebrates were exposed directly to soil. However, no data were available for soil, therefore, it was assumed that mainstem or floodplain sediment (data available) was representative of soil exposure. The exposure assumptions for each receptor and media are presented in Table 8.

Drinking water expected environmental doses (EEDs), which are actually estimated doses, were determined to define exposure of key wildlife species to metals in surface water. Drinking water doses were estimated for all receptors except the fruit bat. The fruit bat derives its water from food; therefore, no dose from water was calculated (Morrison 1980; U.S. EPA 1993). Chronic wildlife water ingestion doses were calculated using Equation (6).

$$EED = \frac{EEC \ x \ WC}{BW} \tag{6}$$

Where:

EED	=	Dose to wildlife species of interest (mg/kg-bw/day)
EEC	=	Expected environmental concentration in water (mg/L)
WC	=	Water consumption rate (L/day)
BW	=	Body weight of receptor (kg-bw)

Table 7. River reaches.

River Reach	Receptors, main stem	Receptors, off river water bodies	Receptors, terrestrial
1	Not assessed for wildlife	Not applicable	Fruit bat, soil invertebrates
2	Not assessed for wildlife	Not applicable	Fruit bat, soil invertebrates
3	Egret, freshwater crocodile, turtles.	Not Applicable	Fruit bat, cassowary, soil invertebrates
4	Egret, freshwater crocodile, turtles.	Egret, White-headed stilt, turtles, freshwater crocodile	Fruit bat, wild pig, cassowary, soil invertebrates
5	Egret, freshwater crocodile, turtles	Egret, White-headed stilt, turtles, freshwater crocodile	Fruit bat, wild pig, cassowary, soil invertebrates
6	Egret, White-headed stilt, estuarine crocodile, turtles	Not Applicable	Fruit bat, Rusa deer, wild pig, cassowary, soil invertebrates
Estuary	Egret, White-headed stilt, estuarine crocodile	Not Applicable	Rusa deer, wild pig, soil invertebrates

Table 8. Exposure media for wildlife receptors.

Wildlife Receptor [a]	Scientific Name	Drinking Water Exposure	Food Exposure and Food Type Assumptions	Incidental Ingestion of Sediment
Fruit Bat	*Dobsonia magna*	No	Fruit	No
Rusa Deer	*Cervus timmorenisis*	Yes	Grass	No
Wild Pig	*Sus sp.*	Yes	Grass	No
Great Egret	*Ardea alba*	Yes	Fish	No
Stilt	*Himantopus leucocephalus*	Yes	Invertebrates	Yes
Cassowary	*Casuarius casuarius*	Yes	Fruit	No
Crocodile, freshwater	*Crocodylus novaguinea*	Yes	Fish	No
Crocodile, estuarine	*Crocodylus porosus*	Yes	Fish	No
Scavenging turtle, freshwater	*Elsya novaegguineae*	Yes	Fish	No
Herbivorous turtle, freshwater	*Caratochelys insculpta*	Yes	Aquatic plants	No

[a] Terrestrial invertebrates were assumed to be exposed directly to soils in flood plain areas.

Water consumption for each species was estimated based on literature values for parameters, such as drinking water rate, or the percentage that direct water consumption contributes to total water intake. The body weights and water consumption rates for wildlife receptors are presented in Table 9. The EEDs for drinking water were based on total rather than dissolved concentrations.

Table 9. Wildlife receptor exposure assumptions.

Receptor	Scientific Name	Body Weight (kg)	Food Ingestion Rate (kg/day)	Water Ingestion Rate (L/day)	References
Fruit bat	*Dobsonia magna*	0.32	0.48	NA	Flannery 1995; Thomas 1984; Funakoshi et al. 1993; Morrison 1980
Rusa Deer	*Cervus timmorenisis*	62.0	19.7	4.1	Woodford and Dunning, 1992; Le Bell et al. 1996
Wild Pig	*Sus sp.*	34.0	5.2	2.4	Professional Judgement; U.S. EPA 1993
Great Egret	*Ardea alba*	0.87	0.22	0.05	Dunning 1993; U.S. EPA 1993
White-headed stilt	*Himantopus leucocephalus*	0.19	0.08	0.02	Dunning 1993; U.S. EPA 1993
Cassowary	*Casuarius casuarius*	44.0	5.9	0.75	Dunning 1993; Reid 1987
Crocodile, freshwater	*Crocodylus novaguinea*	28.5	0.15	1.1	Montague 1984; U.S. EPA 1993
Crocodile, estuarine	*Crocodylus porosus*	10.7	0.07	NA	Taylor 1979; U.S. EPA 1993
Freshwater, Scavenging turtle	*Elsya novaegguineae*	2.0	0.02	0.08	Smith 1998; U.S. EPA 1993
Freshwater, Herbivorous turtle	*Caratochelys insculpta*	15.0	0.62	0.58	Smith 1998; U.S. EPA 1993

NA = Not applicable, obtains water from food

Along with drinking water, metal doses to key receptor species through food ingestion were also evaluated. This was accomplished by using tissue residue data for the appropriate food item or by estimating prey residue concentrations using bioconcentration factors (BCFs) or bioaccumulation factors (BAFs) and then estimating prey consumption. This approach was used even though it is known that BCFs and BAFs have the potential for over estimating prey residue levels. For example, BCFs and BAFs for nutritional metals can be biased towards high values (Chapman 1996). Bias of BCFs and BAFs towards high values occurs when very low water concentrations are used to estimate the BCF (i.e., the organisms were exposed to metal levels that were nutritionally deficient and they subsequently bioconcentrated the metal at a high rate [Chapman 1996]).

All BCFs and BAFs were derived from the scientific literature. BAFs for aquatic invertebrates were from studies where it is known that equilibrium was established. BAFs for terrestrial plants were based on more than one species and plant part, e.g., results for stems and leaves were used. Furthermore, unless noted other wise, all BCFs and BAFs are geometric mean values (Tables 10, 11, and 12).

Prey residue data were unavailable for the white-headed stilt, aquatic plant herbivores, and fruit eating herbivores. Therefore, prey residue concentrations were estimated using BCFs and BAFs (Table 10). Prey residue concentrations were estimated using Equation 7:

$$\text{EERC} = \text{EEC} \times \text{B} \qquad (7)$$

Where:

EERC	=	Expected Environmental Residue Concentration in prey or forage of wildlife receptor (mg/kg)
EEC	=	Expected environmental concentration in water or soil (mg/L or mg/kg)
B	=	Bioaccumulation Factor (BAF) for insect or terrestrial plant or BCF for aquatic plant for each chemical of interest

The chronic wildlife food ingestion doses were calculated using Equation (8). The food items for each receptor are presented in Table 8. The body weights and food consumption rates for wildlife receptors are presented in Table 9.

$$\text{EED} = \frac{\text{EERC} \times \text{FI}}{\text{BW}} \qquad (8)$$

Where:

EED	=	Dose to wildlife species of interest (mg/kg-bw/day)
EERC	=	Expected environmental residue concentration (mg/kg wet weight)
FI	=	Food or forage consumption rate (fish, invertebrates, grass or fruit depending on receptor, kg wet weight/day)
BW	=	Body weight of receptor (kg-bw)

Equation 3-1

Incidental sediment ingestion was only calculated for the white-headed stilt. The assumed incidental sediment ingestion rate was 0.0146 kg/day (U.S. EPA 1993). Chronic wildlife incidental ingestion of sediment doses were calculated using Equation.(9)

$$\text{EED} = \frac{\text{EEC} \times \text{SI}}{\text{BW}} \qquad (9)$$

Where:

EED	=	Dose to white-headed stilt (mg/kg-bw/day)
EEC	=	Expected environmental concentration in sediment (mg/kg)
SI	=	Incidental sediment consumption rate (0.0146 kg/day)
BW	=	Body weight of receptor, (kg-bw)

Table 10. BAFs used for insects and arthropods in wildlife SLRA.

Metal[a]	Insect/Arthropod BAF	References
Aluminium	2,311	Chapman et al. 1968; Sparling and Lowe 1996
Arsenic	9.5	U.S. EPA 1985a
Cadmium	643	U.S. EPA 1996c; U.S. EPA 1985b
Chromium	2,320	Callahan et al. 1979; Chapman et al. 1968
Copper	309	U.S. EPA 1985d
Iron	35	For insects, the value is geometric mean of all nutrient metals (cobalt, copper, manganese, molybdenum, nickel, selenium, vanadium, and zinc)
Lead	748	U.S. EPA 1985e
Manganese	493	U.S. EPA 1992b; PTI 1996
Mercury, inorganic	12,535	U.S. EPA 1996c
Nickel	72	U.S. EPA 1986
Selenium	137	U.S. EPA 1996c
Silver	57	U.S. EPA 1980
Zinc	448	U.S. EPA 1996c; U.S. EPA 1987b; PTI 1996; Callahan et al. 1979

[a] Unless noted otherwise, values are geometric means of values in scientific literature.

Table 11. BCFs used for aquatic plants in wildlife SLRA.

Metal[a]	Aquatic Plant BCF[b]	References
Aluminium	310	Mo et al. 1988; Sparling and Lowe 1996; Thompson et al. 1972
Arsenic	601	Callahan et al. 1979, Chigbo et al. 1982; U.S. EPA 1985a; Jenner et al. 1992; Lindsay and Sanders 1989
Cadmium	353	Callahan et al. 1979, Chandra and Garg 1992, Chigbo et al. 1982; U.S. EPA 1985b, Garate et al. 1993, Garg and Chandra 1993, Huebert and Shay 1991, Muramoto et al. 1989, Nir et al. 1990, Sela et al. 1988; Sinha et al. 1994, Vymazal 1990
Chromium	299	Callahan et al. 1979, Chandra and Garg 1992, U.S. EPA 1985c, Garg and Chandra 1993, Zaranyika et al. 1993, Vymazal 1990,
Copper	817	Chapman et al. 1968; U.S. EPA 1985d; Mo et al. 1988; Sela et al. 1988; Sinha et al. 1994; PTI 1996
Iron	NA	NA
Lead	83	Chapman et al. 1968; Chigbo et al. 1982; Vymazal 1990; PTI 1996
Manganese	76	PTI 1996
Mercury, inorganic	451	Callahan et al. 1979, Chigbo et al. 1982; U.S. EPA 1985f, Lenka et al. 1990
Nickel	891	U.S. EPA 1986; Sela et al. 1988; Muramoto et al. 1989
Selenium	6	PTI 1996
Silver	200	Callahan et al. 1979, U.S. EPA 1987a
Zinc	1,358	Sela et al. 1988

[a] Unless noted otherwise, values are geometric means of values in scientific literature.
[b] When no BCFs were available for terrestrial plants, the geometric mean of all other metals was used.
NA = Not available

Table 12. BAFs for terrestrial plants used in wildlife SLRA.

Metal[a]	Terrestrial Plant BAF[b]	References
Aluminium	0.009	Luwe 1995
Arsenic	0.09	Jenner et al. 1992
Cadmium	0.13	Beyer et al. 1985; Crawford et al. 1995; Lehoczky et al. 1996; Luwe 1995; Moreno et al. 1996; Rabitsch 1995
Chromium	0.002	Jenner et al. 1992
Copper	0.2	Beyer et al. 1985; Jenner et al. 1992; Luwe 1995; Moreno et al. 1996; Rabitsch 1995
Iron	0.06[c]	NA
Lead	0.003	Beyer et al. 1985; Luwe 1995; Jenner et al. 1992; Rabitsch 1995
Manganese	0.06[c]	NA
Mercury, total	0.2	Barghigiani and Ristori 1995; Bull et al. 1977; Cappon 1987; Panda et al. 1992
Nickel	0.08	Jenner et al. 1992; Luwe 1995; Moreno et al. 1996
Selenium	1.7	Jenner et al. 1992; Wu et al. 1995
Silver	0.06[c]	NA
Zinc	0.2	Beyer et al. 1985; Jenner et al. 1992; Luwe 1995; Moreno et al. 1996; Rabitsch 1995

[a] Unless noted otherwise, values are geometric means of literature based values.
[b] Calculate tissue concentration from dry weight concentrations for soil. To convert to wet weight for plants, assume 65 percent water for plant material.
[c] When no BCFs were available for terrestrial plants, the geometric mean of all other metals was used.
NA = Not available.

For each receptor, the food dose was added to the dose computed for the drinking water exposure pathway and incidental ingestion of sediment (if appropriate) to estimate a cumulative dose. In all cases it was assumed that the metals were 100 percent bioavailable (i.e., metals in prey items and surface water are completely bioavailable to wildlife receptors). Assuming 100 percent bioavailability of ingested chemicals is a typical conservative assumption made when conducting screening level risk assessments (U.S. EPA 1997). Other exposure assumptions, such as receptor body weights and food ingestion rates, which are required to estimate chemical intakes (or doses), were derived from the published scientific literature or from U.S. EPA exposure assumption handbooks (e.g., U.S. EPA 1993).

4. EFFECTS CHARACTERISATION

The purpose of the Effects Characterisation is to describe and quantify the potential effects of SOPCs on the various receptors as a function of exposure. The end result of the effects characterisation for the SLRA is development of a series of effect thresholds against which the estimated exposures can be compared. Effects thresholds are typically based on published criteria or guidelines from governmental agencies or, in the absence of such criteria, on information in the scientific literature. The remainder of this section describes the potential effects of SOPCs on the different receptor groups and the rationale behind the effects thresholds that were selected.

4.1 HUMAN HEALTH

The human health effects characterisation defines the toxicological effects thresholds used to estimate risk. The human health effects assessment is divided into two categories, depending on whether or not the substance has potential to cause cancer. Cancer as an effect endpoint is evaluated separately from all other health effects, because cancer-causing chemical stressors are assumed to have potential to cause cancer at any dose (i.e., there is no threshold). Cancer and non-cancer effects are discussed separately below. The thresholds used for non-cancer effects are exposure levels at which no adverse health effects are expected.

Toxicity thresholds are developed to be protective of all individuals in a population including potentially sensitive subpopulations (e.g., children). The toxicity thresholds represent the amount that can be ingested safely on a daily basis for a long period without adverse effects. The potential for adverse effects from short-term (acute) exposure was not directly evaluated because acute effects only occur at dosages much higher than those causing long-term (chronic) effects. Thus, chemicals without predicted chronic effects are also not expected to cause acute effects.

4.1.1 Health Effects other than Cancer

The toxicity thresholds selected were internationally accepted values that are commonly used to assess the potential health significance of chemical exposures (Table 13). Both the World Health Organisation (WHO) and the U.S. Environmental Protection Agency (U.S. EPA) have developed specific methodologies for establishing numerical toxicity thresholds for chemicals causing non-cancerous effects in animals and humans (U.S. EPA 1998b; WHO 1996a).

Toxicity thresholds developed by the WHO for non-cancer health effects are referred to as tolerable daily intakes (TDIs) for ingestion exposures. TDIs are designed to be protective of potentially sensitive subpopulations and of individuals within the general human population. Therefore, exposure to chemicals at or below the TDI is expected to be protective for any exposed individual. Similarly, the U.S. EPA has developed reference doses (RfDs) that are defined as an estimate (with uncertainty spanning perhaps an order of magnitude) of daily exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime (U.S. EPA 1998b). Thus, exposures below the threshold values from each agency may be interpreted similarly (i.e., these stressors do not pose potential risks).

Table 13. Human health toxicity thresholds.

Chemical Stressor	Measurement Endpoint	Oral Threshold Toxicity Value (mg/kg-day)	Reference
Aluminium[a, b]	Aesthetic considerations[a, b]	1	WHO 1996a
Arsenic	Skin changes	0.002	WHO 1996a
Cadmium	Kidney toxicity	0.001	WHO 1996a
Copper	Liver toxicity	0.5	WHO 1996a
Chromium (VI)	Tissue Accumulation	0.003	U.S. EPA 1998b
Iron	Excessive iron build-up in tissue[c]	0.8	WHO 1996a
Lead	Neurological impairment	0.0035	WHO 1996a
Manganese	Neurological effects	0.14	U.S. EPA 1998b
Mercury (Inorganic)	Neurological and kidney effects	0.00071	WHO 1996a
Molybdenum	Elevated serum uric acid	0.005	U.S. EPA 1998b
Nickel	Systemic organ weight changes	0.005	WHO 1996a
Silver	Skin changes	0.005	U.S. EPA 1998b
Zinc [b]	Criterion not health-based[b]	1	WHO 1996a

[a] Based on studies of aluminium phosphate.

[b] WHO has established a provisional maximum tolerable daily intakes for aluminium and zinc that are not based on adverse health effects.

[c] No known adverse health effects are associated with iron accumulation in tissue.

The RfDs and TDIs are available for the ingestion route of exposure, but no toxicity values are currently available for assessing toxicity from the dermal exposure route. Current human health risk assessment methodolgies call for extrapolation of a toxicity value for this exposure route from the ingestion RfD (U.S. EPA 1989a; 1992a). For this risk assessment, oral toxicity thresholds were used to assess exposures from all routes. Generally, threshold toxicity values from the WHO were used because these toxicity values are applicable internationally. In cases where WHO toxicity values were unavailable, U.S. EPA toxicity values were used. In general, WHO and U.S. EPA thresholds are similar, with the exception of arsenic and zinc. The values for these two stressors from U.S. EPA and WHO differ by more than a factor of 10. These differences are accounted for by use of differing safety factors[4] or other methodological differences.

The toxicity thresholds are intended to be very protective. In some cases the uncertainties that arise in the interpretation of either epidemiological or animal data are accounted for using "safety factors." Safety factors are often applied to the experimentally derived toxicity thresholds (thereby reducing their numerical value). They tend to over-estimate potential risk when they are applied (Dourson et al. 1996). Thresholds for nickel and silver were developed using safety factors. Therefore, risks predicted using these thresholds are likely over-estimated. Thus, exposures exceeding a toxicity threshold do not necessarily equate to a health risk. Rather, threshold

[4] Safety factors are used to account for uncertainties in the toxicological data and ensure that toxicity values are adequately protective.

exceedances require further evaluation based on what is known about the stressor and the type and extent of exposure. For some stressors the potential significance of threshold exceedances may be further evaluated by collecting data from tissue (e.g., blood, hair, urine) that will indicate if a person has been exposed. Collection of this type of data from a potentially exposed population may serve as another line of evidence to determine whether excessive exposures may be occurring.

Some chemicals are considered to be essential human nutrients, or are toxic only at high doses. Chemicals that are considered essential nutrients, and that have been evaluated in this risk assessment include copper, iron, manganese, molybdenum and zinc (WHO 1996a,b). If intakes predicted for these chemicals fall within the nutritional concentration range, they have only beneficial effects. The toxicity values for these essential nutrients do not account for the nutritional requirements for these chemicals, therefore, intakes exceeding the toxicity values do not necessarily suggest a potential for adverse effects. For those chemicals that are internally regulated, if internal body concentrations exceed nutritional levels the body will regulate these concentrations to restore them to a normal level. Thus, toxic effects will occur only when doses exceed the body's ability to compensate for any internal concentration changes.

4.1.2 Additive Risks

Additive risks are defined as those resulting from exposure to multiple chemicals that have the same mode of action (Konemann and Pieters 1996; Stara and Erdreich 1984). We reviewed mode of action[5] information for chemicals in Table 13 to identify which have potential for additive effects. We assumed methylmercury and lead could have the same mode of action because they affect the same measurement endpoint (i.e., neurological effects in Table 13), although they may not act by precisely the same means in the nervous system. Although other chemicals shared similar measurement endpoints (e.g., manganese), they did not have the same mode of action and therefore do not have potential for additive risks. No mercury data were available for this SLRA, thus no quantitative evaluation of additive risk potential was possible.

4.1.3 Cancer-Causing Substance: Arsenic

Inorganic arsenic is the only SOPC that has the potential to cause cancer in people by the ingestion exposure route under certain conditions[6]. Ingestion of inorganic arsenic is associated with an increased prevalence of skin cancer, based on epidemiology studies (Tseng 1968, 1977). However, recent reports have called into question the available cancer potency estimates developed using these studies (Abernathy and Roberts 1998, North 1998, Brown 1998). These criticisms are

---

[5] The mode of action refers to the specific way by which a chemical exerts a toxic effect.

[6] Inorganic arsenic has been shown to cause cancer at high doses. However, this does not mean all people exposed will contract cancer. Factors such as metabolism and excretion, exposure duration and other individual factors must be considered.

primarily concerned with two issues: (1) the biologic mechanism of arsenic carcinogenicity and thus the assumed shape of the dose-response curve and risks at low doses; and (2) difficulties in accurately reconstructing doses in the epidemiology studies (Tseng 1968, 1977) used to estimate the slope factor. Assumptions about the mechanism of carcinogenicity are important because, in the absence of information to the contrary, carcinogens are assumed to have no threshold (Wilson 1996). Thus, the dose-response curve is assumed to be linear at doses below those observed in the epidemiology studies, down to very low environmental doses. In cases where the mode of toxic action does not support this assumption, the carcinogenic potency factor would be inappropriately conservative. North (1998) points out that the known modes of action for arsenic would lead to a nonlinear carcinogenic response. A U.S. EPA expert panel has recently concluded that for the biologically plausible mechanisms of arsenic carcinogenicity, "*…the dose-response would either show a threshold or would be nonlinear…*" (U.S. EPA 1998b) (i.e., inorganic arsenic is likely to have a lower carcinogenic potency at low doses). Thus, the linear assumption for arsenic may be inappropriate.

North (1998) states that the Tseng data "…provide a poor basis for estimating the dose-response relationship because the doses to the individuals …are poorly known…". Brown (1998) describes weaknesses in the Tseng data that may lead to the inappropriate association of low doses in the Tseng cohort with skin cancer. U.S. EPA may have underestimated the arsenic dose, thereby overestimating risk, by not accounting for the amount of arsenic in food and the use of arsenic-containing water in food preparation for the Taiwanese population, the study population on which the slope factor is based (Brown and Abernathy 1997).

Based on the preceding discussion, the arsenic cancer slope factor is very likely to result in over-predictions of carcinogenic risk. U.S. EPA has not revised their estimates of the carcinogenic potency of inorganic arsenic at this time and no other international criteria to assess arsenic's carcinogenic potency are available. Therefore, the currently available U.S. EPA slope factor; derived assuming linearity and based on the Tseng (1968, 1977) data, was used. However, it should be interpreted in light of current evidence which suggests that risk predictions using the U.S. EPA value are likely to be over-predicted.

Like all cancer-causing chemicals, the likelihood of contracting cancer is a function of the duration and frequency of arsenic exposure. The endpoint evaluated for arsenic is the probability of contracting skin cancer from ingestion of inorganic arsenic (U.S. EPA 1998b). The toxicity value used to evaluate cancer is the experimentally derived probability of tumour formation as a result of arsenic ingestion. In this assessment the cancer potency factor published by the U.S. EPA (1.5 $(mg/kg\text{-}day)^{-1}$) was used to assess the significance of the cancer causing potential of inorganic arsenic.

4.2 TERRESTRIAL VEGETATION

Similar to the Exposure Characterisation, the effects characterisation for terrestrial vegetation has been divided between physical and chemical stressors, each of which is discussed below.

4.2.1 <u>Effects of Physical Stressors</u>

This subsection first provides a description of the physical stressors to terrestrial vegetation and then develops effects thresholds, largely based on site-specific observational data.

4.2.1.1 Effects of Physical Stressors

The effects of physical stressors vary, depending on the severity of stress and duration of exposure. The adverse effects resulting from exposure to physical stressors ranges from physiological effects on selected species to ecosystem level effects in some locations. This section describes the effects associated with each physical stressor independently, although these effects may be additive in some areas. For example, areas directly affected by sediment deposition also will be exposed to the effects of flooding (but not necessarily vice versa).

Sediment deposition (floodplain aggradation) in terrestrial ecosystems smothers herbaceous and grassy vegetation and can leave a layer of sediment on leaves that inhibits photosynthetic activity. At sufficient depths, soil organisms and ground vegetation are buried and soil respiration is inhibited. The overall result of severe sedimentation is the death of surface vegetation, including all strata (Rau 1994). Secondary effects of vegetation dieback are increased surface temperatures and light penetration, reduced habitat and food for wildlife, and a general loss of ecosystem values such as biological diversity, functional integrity, and nutrient and energy dynamics.

Flooding effects on plants are complex and involve interactions among flood duration, flooding frequency (annual percent flooding), and depth. Plant tolerances to flooding vary with respect to each of these parameters. Some wetland grass species can tolerate prolonged flooding if the upper portion of their shoots remains above the surface. Flood intolerant species, such as many rain forest trees will die as a result of soil waterlogging. The effects of flooding frequency was evaluated in the SLRA because this appears to be the predominant flooding parameter affecting terrestrial vegetation. A thorough discussion of the various flooding parameters and their effects on terrestrial vegetation is provided in Kozlowski (1984).

Flooding can increase stress, causing some plants to respond by increasing aerial roots and/or lenticels on their trunks (Monica Rau, personal communication). While many ground plants (e.g., grasses, forbs, and low shrubs) in the potentially affected vegetation types can withstand complete submersion for short periods, prolonged flooding can result in suffocation and death. Water logging of the soil can result in the death of roots for many terrestrial plant species, particularly in forest vegetation types. Affected trees lose their leaves and fail to produce new ones. Where flooding persists, twigs and branches on affected trees may wither and fall off, and the trees may eventually die. Increased light penetration as a result of the loss of canopy cover may contribute to the chlorosis, which has been observed in understory plants (Duff 1992). Flooding and TSS also cause direct injury through abrasion and flattening of vegetation areas (Duff 1992; Prendergast et al. 1996).

Prolonged flooding can also interfere with soil respiration and other soil processes. Flooding effects on mycorrhizal fungi and other ecologically important soil organisms are not well known; however, they are likely to be adverse.

The effect of scouring is the removal of surface vegetation and erosion of soil on the necks of channel meanders or along channel banks. The effects of scouring can be severe, resulting in the total removal of the biological components of the ecosystem in some areas. However, the spatial extent of scouring effects is estimated to be relatively small in comparison to the effects of floodplain aggradation and flooding frequency (less than 2 percent), and therefore not a significant factor in differentiating among remedial options.

In some locations, all of these physical stressors are operative. Physical damage results from scouring and bank erosion and from flood debris that flattens vegetation. Understory plants are buried by flood debris and sediment, and the increased temperatures and vapour pressure deficit changes microclimatic conditions that adversely affect vegetation (Duff 1992). All areas of floodplain aggradation are within the much broader areas affected by flooding frequency, it is, therefore, difficult to isolate the threshold effects levels for these multiple stressors.

4.2.1.2 Effects Thresholds

For physical stressors, adverse effects are associated with exceedance of normal threshold values for stressor parameters as defined in Section 3.2.2. For example, sediment deposition rates of 1 cm per year may be normal, but a 30 cm per year deposition rate resulting from mine waste disposal may result in adverse effects. Effects thresholds for physical stressors were developed on the basis of observations of effects associated with various levels of the parameters established for each physical stressor.

In the screening process, conservative assumptions are necessary in order to determine threshold effects levels, both at present and in the future. While ecosystem-level assessment endpoints were identified during Problem Formulation, criteria for protection of individuals were used in the SLRA to provide the appropriate degree of conservatism in the screening process.

The screening thresholds are conservative estimates of the stressor levels below which no adverse effects are known to occur. The approach used to develop threshold levels is similar to the approach used for chemical contaminants. Those developed for physical stressors in the screening assessment are generally more conservative than the Lowest Observed Adverse Effects Level (LOAEL) criteria but may not be equivalent to true No Observable Adverse Effects Levels (NOAELs). It was not feasible or appropriate to determine a true and meaningful NOAEL for physical factors because of:

1) The high diversity of the vegetation in exposed ecosystems and limited knowledge of the variability in stressor tolerance limits among plants within a vegetation type (e.g., sedimentation effects on canopy trees versus understory shrubs);

2) The extreme variability in the normal daily, seasonal, annual, and multi-year parameter values of some stressors (e.g., flooding duration);

3) The differences in tolerances among vegetation types to different levels of the physical stressors (e.g., flooding tolerance of foothill forest versus monsoonal savannah); and

4) The fact that each of the physical stressors (i.e., floodplain aggradation, flooding, and scouring) are simply exceedances of the normal range of values for parameters for naturally occurring hydrologic processes in the study area, and these ranges are broad.

In a field investigation, Duff (1992) measured sediment depths (sediment surface to the top of the root zone) at various locations and noted the degree of vegetation stress at each location. His results indicate that no adverse vegetation effects were observed at sediment depths of less than 40 cm. Because of the uncertainties associated with determining the threshold depth, a reasonably conservative threshold depth of 30 cm was adopted for screening purposes.

A threshold value for flooding effects was derived from years of field observations and ongoing investigations of hydrologic parameters and forest dieback effects in the study area. In general, these hydrologic studies document the extreme variability in annual flooding depth between years. Prendergast et al. (1996) adopted an annual flooding value of 30 percent as being that frequency which is sufficient to cause dieback. They adopted this value because: (1) the 1996 flooding frequency at Kuambit is 30 percent where dieback has occurred, and (2) the flooding frequency first remained above 30 percent at Konkonda in 1990; coincidental with the onset of dieback.

The following effects threshold values were developed for use in the SLRA:

Floodplain Aggradation:

- Sediment depth $\geq$ 30 cm

This is less than the shallowest depth associated with observed adverse effects and is therefore appropriate for screening purposes.

Flooding:

- Forested areas (including lowland rain forest): $\geq$30 percent annual inundation. In upper reaches of the Ok Tedi, this value is probably a LOAEL, based on observations of annual flooding and the presence of vegetation that is intolerant to prolonged flooding (Prendergast et al. 1996). In the middle and lower Fly River, the value is probably a conservative NOAEL, based on the flood tolerance of vegetation in these reaches (Bourliere 1983b; Sekhran and Miller 1995; Prendergast et al. 1996). Prendergast et al. (1996) report a pre-mine floodplain inundation rate of 10 percent at Konkonda and 55 percent at Obo. These rates can be considered NOAELs for the respective portions of the study area and indicate the gradient of percent flooding that exists between the upper and lower reaches of the Ok Tedi and Fly River systems.

- Grassland savannahs: $\geq$60 percent annual inundation. This value is based on field measurements made by OTML investigators and others within the study area (Prendergast et al. 1996) and is supported by statements in Bourliere (1983b) who noted that seasonal inundation for a period of seven months is a prominent feature of the savannahs of southwestern PNG.

As a conservative assumption, floodplain elevation rather than bank elevation was used as the default level indicative of flooding. While the higher banks form natural levees, these are sometimes breached resulting in flooding of the adjacent floodplain. The values of 30 cm depth selected as a screening threshold for aggradation and 30 percent annual inundation for flooding are appropriately conservative, given the natural variability of the environment, particularly as they were developed on the basis of site-specific information.

Scouring:

- Qualitative assessment, based on personal observation and evaluation of aerial photography for reaches where aggradation and flooding effects from mine waste disposal are known or expected. Scouring is likely an insignificant contributor to overall vegetation/habitat loss.

Because mine waste discharge will continue into the future, modelling the effects of sedimentation and consequent flooding on the river system provided a means of evaluating potential adverse effects for each river reach. Predictions of floodplain aggradation (sedimentation) and related flooding for screening purposes were obtained from a mathematical morphological model of the Ok Tedi-Fly River system (Klohn-Crippen 1996). The model was developed to assess future physical and biological effects at various nodes in order to estimate the length of time it would take the river conditions to be restored to 1985 levels. Model outputs for floodplain aggradation included estimates of floodplain aggradation (depth) from 1991 through 2050. Model outputs for flooding frequency were estimates of the percent annual flooding (flooding frequency) in relation to both the bank (levee) elevation and floodplain elevation, from 1991 through 2050. As the floodplain level was the lower of the two, it was selected for screening purposes as the more conservative estimate of flooding frequency. If estimates of floodplain aggradation or flooding frequency exceeded threshold levels in a river reach at any time during the period of prediction, the reach and stressor were retained for further evaluation in the DLRA.

A more sophisticated hydrologic model was developed by Cui and Parker after the draft SLRA had been completed. Comparisons of physical thresholds with outputs of this model produced the same screening results as the Klohn-Crippen model. The Cui-Parker model was used for subsequent analyses in the DLRA.

Both models indicate that both floodplain aggradation and flooding effects resulting from mine waste discharge, including the secondary effects of scouring, will continue for at least 50 more years.

4.2.2 Effects of Chemical Stressors

4.2.2.1 Effects of Divalent Metals

Cadmium

Cadmium is chemically similar to zinc, an essential element, and competition may occur between the two for organic ligands. This may explain some of the ameliorative effects of zinc on cadmium phytotoxicity. Cadmium depresses uptake of iron, manganese, and probably calcium, magnesium, and nitrogen (Efroymson et al. 1997). Phytotoxic symptoms resemble iron chlorosis and include reduced growth, wilting, reduced zinc levels, and necrosis. The mechanisms of phytotoxicity include reduced photosynthetic rate, poor root system development, reduced conductivity of stems, and ion interactions in plants (Efroymson et al. 1997). Adriano (1986) has suggested that agronomic crops are more sensitive to cadmium than trees. Efroymson et al. (1997) provided phytotoxicity-based soil thresholds for cadmium of 4 mg/kg and 0.1 mg/L in soil solution. Cureton et al. (1994) indicated that cadmium reduced lettuce and radish root elongation in 50 percent of test plants at concentrations of 3 and 38 mg/kg, respectively. Seedling emergence by these two species was less sensitive, with EC_{50} values of 205 and 143 mg/kg, respectively.

<u>Copper</u>

Copper is essential to a number of plant functions, including oxidation, photosynthesis, and protein and carbohydrate metabolism, and is potentially involved in symbiotic nitrogen fixation, valence changes, and cell wall metabolism (Kabata-Pendias and Pendias1992). Excess copper interferes with enzyme function in the root system, photosynthesis, and fatty acid synthesis.

Excess copper can inhibit nitrogen uptake by plants, particularly nitrate, and decrease the content of nitrate, amino acids, and proteins in plants (Weber et al. 1991). Cureton et al. (1994) identified EC_{50} values for seedling emergence by lettuce and radish of approximately 90 mg/kg, and EC_{50} values effecting root elongation of lettuce and radish of 3 to 13 mg/kg, respectively. Efroymson et al. (1997) provided phytotoxicity-based soil thresholds for copper of 100 mg/kg and 0.06 mg/L in soil solution. CCME (1997a,b) provided a soil quality threshold for copper of 63 mg/kg. They indicated that the lowest reported copper concentration that produced phytotoxic effects was 50 mg/kg in a dry soil, resulting in 18 percent decrease in radical elongation in Paper Birch (*Betula papyrifera*). However, in the same study, 50 mg/kg produced no observed adverse effect in other plant species (*Pinus* sp.), indicating inter-species sensitivity to copper. Alloway (1995) indicated that inter-species (and inter-cultivar) sensitivity of plants to copper exists, and it is difficult to establish a single concentration associated with copper phytotoxicity for all plants.

<u>Lead</u>

The phytotoxicity of lead is relatively low compared with other trace elements. Excess lead affects mitochondrial respiration and photosynthesis by disturbing electron transfer reactions (Efroymson et al. 1997). Efroymson et al. (1997) have provided phytotoxicity-based soil thresholds for lead of 50 mg/kg (dry wt.) and 0.02 mg/L in soil solution. Cureton et al. (1994) indicated that lead reduced lettuce and radish root elongation in 50 percent of test plants at concentrations of 35 and 60 mg/kg, respectively. Seedling emergence by these species was less sensitive, with an EC_{50} of 900 mg/kg for both species. CCME (1997b) provided a soil quality threshold for lead of 70 mg/kg for agricultural soils and 140 mg/kg in soils used for residential or parkland uses.

Nickel

Nickel is not considered an essential element for plant nutrition, but it may be required for modulated legumes for internal nitrogen transport as part of the urease enzyme (Efroymson et al. 1997). Nickel is generally absorbed as the Ni (II) ion and translocated in xylem and phloem with an organic chelate. Nickel is uniformly distributed between roots and shoots. Symptoms of nickel phytotoxicity include iron deficiency-induced chlorosis, poor growth, and foliar necrosis (Collins 1981). Excess nickel affects nutrient adsorption by roots, root development and metabolism, and inhibits photosynthesis and transportation. Nickel can replace cobalt and other metals located at active sites in metallo-enzymes and disrupt their functioning (Efroymson et al. 1997). Efroymson et al. (1997) have provided phytotoxicity-based soil thresholds for nickel of 30 mg/kg (dry wt.) and 0.5 mg/L in soil solution.

Zinc

Zinc is an essential element to plants, is important in many enzymes, and is involved in disease protection and metabolism of carbohydrates and proteins (Efroymson et al. 1997). Zinc acts by altering membrane permeability and inhibiting CO_2 fixation and phloem transport of carbohydrates.

Efroymson et al. (1997) provided phytotoxicity-based soil thresholds for zinc of 50 mg/kg (dry wt.) and 0.4 mg/L in soil solution. Cureton et al. (1994) indicated that zinc reduced lettuce and radish root elongation in 50 percent of test plants at concentrations of 16 and 38 mg/kg, respectively. CCME (1997b) indicated that the lowest reported zinc concentration producing phytotoxic effects was 50 mg/kg, which resulted in seed yield reduction in turnip at pH 6.3. Zinc interacts with other nutrients in soil, and iron deficiency may occur when excess zinc is present (CCME 1997b). CCME (1997b) provided soil quality thresholds for zinc for agricultural, parkland, and residential uses of 200 mg/kg.

4.2.2.2 Effects of Metals with Organic Forms

Arsenic

The phytotoxicity of arsenic is influenced by the form in the soil (Marin et al. 1992). Arsenic (III) is more toxic than arsenic (V) (Peterson et al. 1981). Rice and legumes appear to be more susceptible to arsenic than other plants (Efroymson et al. 1997).

Because arsenic is chemically similar to phosphorus, it is translocated in the plant in a similar manner and is able to replace phosphorus in many cell reactions. Arsenite probably reacts with sulphydryl enzymes leading to membrane degradation and cell death (Peterson et al. 1981). Arsenate is known to uncouple phosphorylation and affect enzyme systems (Peterson et al. 1981). Arsenate interferes with protein synthesis, and/or the phosphorylation of proteins, within plants, leading to phytotoxic symptoms.

CCME (1997b) provides a soil quality threshold for arsenic of 1.4 mg/kg for agricultural soils and 10 mg/kg for residential/parkland land uses. The lowest reported soil arsenic concentration identified that produced a phytotoxic effect was 10 mg/kg. Depending on soil type, this concentration resulted in yield reductions of 22 and 42 percent in green beans, 33 and 41 percent in spinach, and 17 and 23 percent in radish (Woolson 1973). Yield reductions of 26 percent in cabbage and 22 percent in lima beans have also been observed at 10 mg/kg. In the same study, tomatoes were not affected by 100 mg/kg.

Cureton et al. (1994) provide concentrations (EC_{50}) affecting seedling emergence of lettuce and radish of 14 and 46 mg/kg, respectively, and EC_{50} values affecting root elongation of lettuce and radish of 1 and 12 mg/kg, respectively. Efroymson et al. (1997) has provided phytotoxicity-based soil thresholds for arsenic of 10 mg/kg (dry wt.) and 0.001 mg/L in soil solution. CCME (1997b) provided a soil quality threshold for inorganic arsenic (total) of 12 mg/kg.

Mercury

Literature sources indicate that a number of growth endpoints are reduced by 50 percent at concentrations ranging from 7 to 1000 mg/kg (EC 1995; Sheppard at al. 1993). Efroymson et al. (1997) have provided phytotoxicity-based soil thresholds for mercury (total) of 0.3 mg/kg (dry wt.) and 0.005 mg/L in soil solution. They also provided a phytotoxicity-based soil threshold for methylmercury of 0.0002 mg/L in soil solution. Cureton et al. (1994) indicated that mercury (total) reduced lettuce and radish root elongation in 50 percent of test plants at concentrations of 18 and 70 mg/kg, respectively. Seedling emergence by these species was less sensitive, with EC_{50} values of 103 and 15 mg/kg, respectively (Cureton et al. 1994). Reductions of 25 percent in seedling emergence of lettuce and radish occurred at 11 and 73 mg/kg, respectively (EC 1995).

Selenium

Terrestrial plants in many areas of the world contain concentrations of selenium that are deficient or toxic relative to animal metabolism. When plants drop their leaves or die and decay, readily absorbable selenium is increased in the root area of adjacent plants. Many plants accumulate selenium concentrations greater than the growth media, and plants may be categorised into hyperaccumulators (>1000 mg/kg); secondary accumulators (>100 mg/kg); and non-accumulators (<30 mg/kg) (Peterson et al. 1981).

A mechanism of selenium toxicity in terrestrial plants may be due to indiscriminate replacement of sulphur by selenium in proteins and nucleic acids with disruptions in metabolism (Trelease et al. 1960). Selenomethionine is a less effective substrate than methionine for peptide bond formation, which could reduce protein synthesis.

4.2.2.3 Effect of Other Chemical Stressors

Aluminium

Aluminium can interfere with cell division in plant roots; decrease root respiration; fix phosphorus in unavailable forms in roots; interfere with uptake, transport and use of calcium, magnesium, phosphorus and potassium and water; and interfere with enzyme activities (Foy et al. 1978).

Seedlings are more susceptible to damage from aluminium toxicity than are older plants (Efroymson et al. 1997).

Efroymson et al. (1997) provided phytotoxicity-based soil thresholds for aluminium of 50 mg/kg (soil pH 5.0) and 0.3 mg/L in soil solution. Seedling establishment by white clover in a silt loam soil (pH 5.0) has been reduced by 30 percent by addition of 50 mg/kg aluminium (Efroymson et al. 1997). However, because aluminium is a constituent of clay minerals, toxicity to plants is theoretically possible in most, if not all soils when the soil pH decreases to levels low enough to cause the clay mineral structure to decompose, approximately pH of <5.5 (Petersen and Girling 1981; Foy et al. 1978). Ahlrichs et al. (1990) have undertaken root elongation tests on 243 soil samples and found that approximately half the samples with pH ≤5.0 showed aluminium toxicity. However, Ahlrichs et al. (1990) indicated that some soils with pH ≤5.0 and elevated aluminium in the soil solution did not always show aluminium toxicity, indicating that the species of aluminium in the soil was important for aluminium toxicity.

In acid soils, some of the aluminium, formerly a part of the clay particles, migrates to cation exchange sites on clay surfaces and into the soil solution. Raising the pH of the solution above 5.5 usually precipitates aluminium and negates the phytotoxicity (Petersen and Girling 1981).

Chromium

Chromium is not an essential element in plants. Cr (VI) form is more soluble than Cr (III), and is considered the more phytotoxic form (Efroymson et al. 1997). In soils with normal Eh and pH range, Cr (VI), a strong oxidant, is likely to be reduced to the less available Cr (III). However, the Cr (III) form may be oxidised to the Cr (VI) form in the presence of oxidised manganese (Efroymson et al. 1997).

Cr (VI) is more mobile in plants that Cr (III), but translocation varies with plant type (Efroymson et al. 1997). After plant uptake, chromium generally remains in the roots. Within the plant, Cr (VI) may be reduced to Cr (III) and complexed as an anion with organic molecules (Efroymson et al. 1997).

Efroymson et al. (1997) have provided phytotoxicity-based soil thresholds for chromium of 1 mg/kg (dry wt.) and 0.05 mg/L in soil solution; however, confidence in the whole-soil threshold value is low as natural background levels of chromium in sediments along the river exceed the threshold (UNEP, 1995). UNEP (1995) have indicated that the estimated natural concentration of chromium (total) in sediments at Reach 4 was 61 ±16 mg/kg (mean ±SD).

Cureton et al. (1994) indicated that Cr (III) reduced lettuce and radish root elongation in 50 percent of test plants at concentrations of 11 and 33 mg/kg, respectively, and 3 and 26 mg/kg, respectively, for Cr (VI). CCME (1997b) provided a soil quality threshold for chromium (total) of 64 mg/kg. CCME (1997b) indicated that the lowest reported total chromium concentration resulting in phytotoxic effects was 21 and 31 mg/kg, which caused a 50 percent decrease in yield of tomato and oats, respectively. Radish and lettuce seed germination was reduced by 50 percent at concentrations of 81 and 397 mg/kg, respectively (CCME 1997b). Phytotoxic effects of Cr (VI) have been recorded at 1.8 and 6.8 mg/kg, resulting in a 50 percent yield reduction of lettuce and tomato, respectively (Adema and Henzen 1989).

Iron and Manganese

Iron and manganese are essential plant nutrients. Iron and manganese oxides play an important role in the soil in fixing trace elements such as cobalt, copper, zinc, lead and nickel (Norrish 1975). The association of these elements with iron and manganese has important implications for plant growth. Studies have shown that fixation of elements by iron and manganese is rapid and results in the elements being unavailable to plants.

The availability of manganese to plants is dependent on oxidising conditions and pH. Under acidic conditions, manganese is sufficiently soluble but manganese deficiencies may occur with a pH of 7 to 8. Efroymson et al. (1997) provided a soil quality threshold for manganese of 500 mg/kg and 4 mg/L in soil solution.

Iron solution concentrations of 10 to 50 mg/L have produced adverse growth effects in plants in laboratory bioassays (Efroymson et al. 1997). Solution-culture experiments have indicated differential tolerance to iron (as Fe [II]) exhibited by wetland plant species (e.g., iron-tolerant and iron-susceptible wetland plant species), and those showing greater tolerance have the capacity to form hydrated iron oxide (ochre) root precipitates (Snowden and Wheeler 1995). Monocots are generally more tolerant of Fe (II) than dicots, and iron-sensitive wetland plants are confined to wetland sites with comparatively low iron availability (Snowden and Wheeler 1995).

Oxidative precipitation of iron on roots is an important iron-tolerance mechanism in wetland plants; however, wetland plant species differ in their capacity to produce ochre, and iron-tolerant species tend to form ochre more readily. Monocots, which are predominant in many wetlands, produce ochre on their roots more readily than dicots (Snowdon and Wheeler 1995). Ochre formation is not just species or cultivar-dependent, as some plants may form ochre in the absence of phosphorus, but will form an iron-phosphorus precipitate in the presence of phosphorus, and thus ochre formation may be environment-dependent (Snowdon and Wheeler 1995).

Iron toxicity to plants is a complex issue and the effects of iron may be both direct and indirect, due to exclusion of uptake of plant nutrients (Snowdon and Wheeler 1995). Effects may be positive or negative. Yield reduction in exposed plants has been linked directly to iron uptake. Root oxidative iron-precipitation may help to keep iron uptake by plants under threshold concentrations, reported to be 0.3 mg/g for rice. There is no evidence to indicate that iron-tolerant species can cope with higher tissue concentrations of iron than iron-sensitive species, rather it is the capacity for iron exclusion that is probably more important (Snowdon and Wheeler 1995).

Molybdenum

Molybdenum is required by plants for symbiotic nitogen fixation by legumes and for growth of non-leguminous plants. Molybdenum is important for enzymes active in nitrogen metabolism (activation of nitrogenase and nitrate reductase). Phytotoxic symptoms include chlorosis, apparently due to interference with iron metabolism (Efroymson et al. 1997). Efroymson et al.

(1997) provided a phytotoxicity-based soil threshold for molybdenum of 2 mg/kg, and 0.2 mg/L in soil solution.

<u>Silver</u>

The phytotoxicity of silver is related to the binding potential of Ag (I) ions to enzymes and other active molecules at cell surfaces (Cooper and Jolly 1970). Binding of -SH groups in the formation of mercaptides is the principal mechanism of enzyme inhibition. Phytotoxicity is reduced in the presence of organic matter (U.S. EPA 1981). Efroymson et al. (1997) reported a lack of phytotoxicity data from plants grown in soils, but provided a phytotoxicity-based soil threshold for silver in soil solution of 0.1 mg/L. No phytotoxicity threshold is available for silver (total) in whole soils.

<u>Natural Background Sediment Quality</u>

Natural background sediment quality data have been reported by UNEP (1995) from Reach 4. Samples were collected in the early 1990's using a 36-mm diameter gravity corer with an acrylic liner. Background sediment samples were taken from the lower portions of the cores, beneath the mine-derived sediments. Samples were wet-sieved with the <20µm fraction saved for digestion in aqua regia and chemical analysis.

The UNEP data are presented in Table 14. The elements Al>V>Co>Cr>Sc>Ni>Ti>Zr (in decreasing order) are present in mine-related materials (waste rock and tailings) in lower concentrations than in background sediments at Reach 4 (UNEP 1995). As such, the presence of these analytes in mine-related wastes is unlikely pose an unacceptable risk to the environment. Some level of uncertainty is added to the comparisons of background sediments with mine-related sediments because of two factors: 1) unknown collection and processing methods associated with the OTML (1996) and the CSIRO (1998) mine-derived sediment data, and 2) comparisons of bulk sediment results without consideration of grain size, organic matter, or other factors that could affect the form and mobility of the metals. However, comparisons of contaminant concentrations in sediment with phytotoxicity thresholds are based on bulk sediment non-normalized concentrations.

Table 14. Background analyte concentrations in Reach 4 (Middle Fly River) sediments.

Analyte	Reach 4 (Middle Fly River) Mean Background Sediment Concentration (mg/kg dw)	Standard Deviation (n = 128 samples)	95% UCL of Mean
Aluminium	86000	25000	89000
Arsenic	4.6	3.6	5.1
Cadmium	0.26	0.13	0.3
Chromium	61	16	63
Cobalt	13	2	-
Copper	45	19	48
Iron	37000	15000	39000
Lead	18	9	19.3
Manganese	290	225	321
Molybdenum	1.7	1.4	1.9
Nickel	33	13	35
Scandium	17	5	-
Silver	0.24	0.26	0.3
Titanium	3765	1269	-
Vanadium	165	49	-
Zinc	140	54	150
Zirconium	67	25	-

Source: Table 8 in UNEP (1995).

4.2.2.4 Effects Thresholds

Canada is the only country to have completed development of soil quality thresholds (CCME 1997a,b) for the protection of terrestrial vegetation, although several additional countries are in the process of developing thresholds. Other sources of phytotoxicity-based soil thresholds include Efroymson et al. (1997). Table 15 presents phytotoxicity-based soil thresholds used in this assessment.

An important caveat to these thresholds is their applicability in environments outside those in which they were developed. Many regions in the world naturally have high levels of mineralisation, and the plant species in these regions have adapted through evolutionary change to these natural conditions. The phytotoxicity-based thresholds used in this assessment are based on data for plant species that are not adapted to highly mineralised conditions. Accordingly, natural background data is normally used to evaluate the validity of threshold values. Based on the background sediment data available (UNEP, 1995), all phytotoxicity-based thresholds were greater than the estimated 95 percent UCL of the mean of the background analyte concentrations at Reach 4, except for chromium, nickel, and zinc. UNEP (1995) data indicated 95 percent UCL natural

Table 15. Proposed phytotoxicity-based soil thresholds for terrestrial vegetation.

Chemical	Toxicity Threshold	Reference

Table 15. Proposed phytotoxicity-based soil thresholds for terrestrial vegetation (continued).

	(mg/kg dw)	
Aluminium	89000	UNEP (1995) 95% UCL of Estimated Background Sediment Concentration in Reach 4
Arsenic	10	Efroymson et al. 1997; CCME 1997b
Cadmium	4	Efroymson et al. 1997
Chromium (total)	63	UNEP (1995) 95% UCL of Estimated Background Sediment Concentration in Reach 4
Copper	63	CCME 1997b
Iron	39000	UNEP (1995) 95% UCL of Estimated Background Sediment Concentration in Reach 4
Lead	50	Efroymson et al. 1997
Manganese	500	Efroymson et al. 1997
Molybdenum	2	Efroymson et al. 1997
Nickel	35	UNEP (1995) 95% UCL of Estimated Background Sediment Concentration in Reach 4
Silver	2	Efroymson et al. 1997
Zinc	150	UNEP (1995) 95% UCL of Estimated Background Sediment Concentration in Reach 4

background concentrations of chromium, nickel, and zinc of 61 mg/kg, 35 mg/kg, and 142 mg/kg, respectively at Reach 4. Phytotoxicity-based thresholds available for chromium, nickel, and zinc identified for this assessment were 21 mg/kg, 30 mg/kg, and 50 mg/kg. Thus, the phytotoxicity-based thresholds are considered too conservative for these analytes and the background concentrations recorded by UNEP (1995) have been adopted for evaluation of sediment data for these SOPCs.

No phytotoxicity-based thresholds were available for aluminium and iron; however, the estimated natural background concentrations of these analytes from Reach 4 (89000 mg/kg and 39000 mg/kg, respectively) have been adopted from UNEP (1995) for use as phytotoxicity-based thresholds.

The chemistry and phytotoxicity of analytes in sediments may, in general, differ from that of soils that are not permanently or predominantly waterlogged; however, no phytotoxicity-based sediment quality thresholds were available for this assessment. The sediments on the riverbanks, levees and floodplains along the river system undergo cycles of flooding and drying. Thus, the phytotoxicity-based soil guidelines are considered applicable for assessing the risks to terrestrial plants under the various soil moisture conditions along the river system.

In general, the metal bioavailability, and thus the potential for phytotoxicity, is probably a function of the metal concentration in the soil or sediment solution (i.e., the interstitial water concentration). Whole-soil or whole-sediment analyte concentrations may be less predictive of phytotoxicity than soil/sediment solution analyte concentrations due to various parameters (e.g., organic matter content, clay content, pH). Phytotoxicity-based soil solution thresholds were available for a range of analytes for this assessment (i.e., Efroymson et. al. 1997); however, only whole-sediment analytical data were available from the study area.

4.3 AQUATIC LIFE

Since the purpose of the SLRA is to evaluate aquatic communities along the various river reaches, surface water and sediment criteria/guidelines designed to protect aquatic communities were used to assess potential effects. The sources and types of criteria/guidelines selected for surface water and sediment are described below.

4.3.1 Surface Water

The effects of SOPCs on aquatic life are divided into chemical and physical stressors for discussion purposes in this section.

4.3.1.1 Chemical Stressors

Water quality criteria were identified that are intended to protect all of, or 95 percent of, the aquatic community. As explained in the exposure characterisation, both acute and chronic exposures were evaluated. Of the primary water quality criteria sources listed below, only the U.S. EPA publishes acute criteria. Both freshwater and saltwater criteria were identified. For the SLRA, freshwater criteria were used for Reaches 1 through 5 and saltwater criteria were used for Reach 6 and the estuary. The primary sources of water quality criteria were:

- Papua New Guinea (PNG 1998);
- Australian and New Zealand Environment Conservation Council (ANZECC 1992);
- United States Environmental Protection Agency (U.S. EPA); and
- Canadian Council of Ministers of the Environment (CCME 1995 plus updates).

With the exception of molybdenum, water quality criteria were available for all of the metals and metalloids identified for evaluation in the SLRA. Aquatic toxicity data for molybdenum are limited, but Kimball (1978) studied its acute and chronic effects on the freshwater cladoceran *Daphnia magna*. Because appropriate toxicity data are limited to one species, an uncertainty factor of 21.9 was applied to both the acute and chronic values from the Kimball (1978) study to account for the possibility that there are other freshwater species more sensitive to molybdenum than *D. magna*. The uncertainty factor of 21.9 is recommended by the U.S. EPA (1995) when data are available for only one species[7]. Additionally, because criteria for manganese are limited, acute and chronic manganese criteria derived by Parametrix (1997) were included. Parametrix reviewed the scientific literature for manganese toxicity data using the same procedure that the U.S. EPA uses (Stephan et al. 1985) in deriving water quality criteria. As with other divalent metals, manganese

[7] The U.S. EPA recommends increasingly greater uncertainty factors as the number of species for which there is toxicity data decreases.

toxicity is hardness-dependent, and manganese criteria are expressed using hardness-dependent equations as described in U.S. EPA guidance (Stephan et al. 1985). The criteria for chemical stressors in freshwater and salt water are presented in Tables 16 and 17, respectively.

The guidelines published by Papua New Guinea, ANZECC, and the CCME are expressed as total recoverable metal, and in their original development, the U.S. EPA criteria are also expressed as total recoverable metal. However, since as early as the mid-1970s it has been demonstrated that the dissolved fraction of most metals more closely represents the bioavailable form to aquatic life (e.g., Meador 1991; Nelson et al. 1986; Pagenkopf 1983; Part and Svanberg 1981; Sunda and Guillard 1976; Zamuda and Sunda 1982).

In this document, in discussing the exposure and effects data used in the risk assessment, the term 'dissolved' is operationally defined as the amount of metal passed through a 0.45 µm filter. Dissolved metal only approximates bioavailable metal because metals binding to non-bioavailable organic colloids of ligands may also pass through a 0.45 µm filter. The U.S. EPA has estimated the fraction of various metals and metalloids that are dissolved in the toxicity tests on which the criteria are based (U.S. EPA 1996a). These fractions, termed conversion factors, can be directly applied to the criteria expressed as total recoverable metal to estimate what the criteria are on a dissolved basis. Although these factors were developed based on the toxicity tests used to develop U.S. EPA criteria, the factors were also applied to criteria from the other countries. This was assumed to be a reasonable approach since the toxicity databases on which the various criteria are based overlap considerably with the U.S. EPA databases. The conversion factors for estimating dissolved criteria are shown in Table 18 below. For metals without dissolved conversion factors (i.e., aluminium, manganese, and molybdenum), the dissolved fraction could not be estimated and criteria were expressed as total recoverable metal. Given how close to one most of the conversion factors for other metals are, this is not expected to significantly influence the risk estimations (although dissolved aluminium may be overestimated given its low solubility at circumneutral pH [U.S. EPA 1988a]).

4.3.1.2 Physical Stressors

None of the primary sources for chemical criteria have published TSS criteria. Consequently, TSS effects data published in the scientific literature were reviewed to identify suitable screening criteria for the SLRA. Sufficient acute toxicity data for fish were available to perform a logistic regression, which was used to estimate the TSS concentration that would protect 95 percent of the fish species (Table F-1 in Appendix F). This resulted in an acute criterion of 405 mg/L. Chronic TSS effects data were more limited, so it was not possible to perform a logistic regression. The lowest chronic value was a No Observable Effect Concentration (NOEC) of 270 mg/L for mortality in rainbow trout (EIFAC 1965) (Table F-2 in Appendix F). Vascular plants appear to be more sensitive than fish, with Lowest Observable Effect Concentrations (LOECs) as low as 50 mg/L for reduced growth relative to controls (Otto and Enger 1960) (Table F-3 in Appendix F). These values were used to evaluate TSS along the entire length of the Fly River.

Table 16. Surface water criteria for chemical stressors in freshwater (µg/L).

	PNG		ANZECC		U.S. EPA		CCME		Other	
Analyte	Acute	Chronic	Acute	Chronic	Acute	Chronic	Acute	Chronic	Acute	Chronic
Aluminium	---	100	---	100	750	87	---	100	---	---
Arsenic	---	50	---	50	339.8	147.9	---	50	---	---
Cadmium	---	$e^{(0.7852*\ln(Hard.)-3.49)}$	---	0.2 - 2.0	$e^{(1.128*\ln(Hard.)-3.6867)}$	$e^{(0.7852*\ln(Hard.)-2.715)}$	---	$10^{(0.86*Log(Hard.)-3.2)}$	---	---
Calcium	---	---	---	---	---	---	---	---	---	---
Chromium	---	10	---	10	16	11	---	2	---	---
Copper	---	$e^{(0.8545*\ln(Hard.)-1.465)}$	---	2.0 - 5.0	$e^{(0.9422*\ln(Hard.)-1.7)}$	$e^{(0.8545*\ln(Hard.)-1.702)}$	---	2 - 4	---	---
Iron	---	1,000	---	1,000	---	1,000	---	300	---	---
Lead	---	$e^{(1.273*\ln(Hard.)-4.705)}$	---	1.0 - 5.0	$e^{(1.273*\ln(Hard.)-1.46)}$	$e^{(1.273*\ln(Hard.)-4.7)}$	---	1 - 7	---	---
Manganese	---	500	---	---	---	---	---	---	$e^{(0.6687*\ln(Hard.)+5.076)}$	$e^{(0.6687*\ln(Hard.)+4.121)}$
Mercury	---	0.1	---	0.1	1.694	0.9081	---	0.1	---	---
Molybdenum	---	---	---	---	---	---	---	---	4,721	40
Nickel	---	$e^{(0.76*\ln(Hard.)+1.06)}$	---	15 - 150	$e^{(0.846*\ln(Hard.)+2.255)}$	$e^{(0.846*\ln(Hard.)+0.0584)}$	---	25 - 150	---	---
Selenium	---	5	---	5	19.34	5	---	1	---	---
Silver	---	0.1	---	0.1	0.92	0.12	---	0.1	---	---
Sulphate	---	400,000	---	---	---	---	---	---	---	---
Zinc	---	$e^{(0.83*\ln(Hard.)+1.95)}$	---	5 - 50	$e^{(0.8473*\ln(Hard.)+0.884)}$	$e^{(0.8473*\ln(Hard.)+0.884)}$	---	30	---	---

Table 17. Surface water criteria for chemical stressors in salt water (µg/L).

	PNG		ANZECC		U.S. EPA		CCME		Other	
Analyte	Acute	Chronic	Acute	Chronic	Acute	Chronic	Acute	Chronic	Acute	Chronic
Aluminium	---	---	---	---	---	---	---	---	---	---
Arsenic	---	50	---	50	69	36	---	---	---	---
Cadmium	---	2	---	2	43	9.3	---	0.1	---	---
Calcium	---	---	---	---	---	---	---	---	---	---
Chromium	---	50	---	50	1,100	50	---	---	---	---
Copper	---	5	---	5	4.8	3.1	---	---	---	---
Iron	---	---	---	---	---	---	---	---	---	---
Lead	---	4	---	5	220	5.6	---	---	---	---
Manganese	---	2,000	---	---	---	---	---	---	---	---
Mercury	---	0.1	---	0.1	2.1	1.1	---	---	---	---
Molybdenum	---	---	---	---	---	---	---	---	---	---
Nickel	---	15	---	15	75	8.3	---	---	---	---
Selenium	---	70	---	70	300	71	---	---	---	---
Silver	---	1	---	1	7.2	0.92	---	---	---	---
Sulphate	---	---	---	---	---	---	---	---	---	---
Zinc	---	50	---	50	95	86	---	---	---	---

how close to one most of the conversion factors for other metals are, this is not expected to significantly influence the risk estimations (although dissolved aluminium may be overestimated given its low solubility at circumneutral pH [U.S. EPA 1988a]).

Table 18. Conversion factors for estimating dissolved metals from total recoverable.

	Freshwater		Saltwater
Metal	Acute	Chronic	Acute
Aluminium	NA	NA	NA
Arsenic	1.000	1.000	1.000
Cadmium	Hardness-dependent[a]	Hardness-dependent[a]	0.994
Chromium(VI)	0.982	0.962	0.993
Copper	0.960	0.960	N/AP[b]
Lead	Hardness-dependent[a]	Hardness-dependent[a]	0.951
Manganese	NA	NA	NA
Mercury	0.85	NA	0.85
Molybdenum	NA	NA	NA
Nickel	0.998	0.997	0.990
Selenium	NA	NA	0.998
Silver	0.85	NA	0.85
Zinc	0.978	0.986	0.946

[a] Cadmium Acute: 1.136672-ln(hardness)(0.041838)
Chronic: 1.101672-ln(hardness)(0.041838)
Lead Acute and chronic: 1.46203-ln(hardness)(0.145712)
[b] The saltwater copper acute criterion is already expressed as dissolved metal.
NA = Not available (the U.S. EPA has not identified conversion factors for these metals)
N/AP = Not applicable

It is recognised that these thresholds are based largely on data for North American species and that aquatic life in the Ok Tedi/Fly River system will be naturally more tolerant of TSS than these species. This issue will be addressed to the extent data are available in the DLRA. Additionally, it is recognised that these literature-based values are likely below background (i.e., pre-mine) concentrations for the Fly River below the Strickland. In the DLRA, using historical data and data for similar rivers, TSS will be evaluated further by identifying fish communities that inhabited the different river reaches before mining commenced. A rank scoring method will be used to assess the potential effects of TSS on various fish species.

Aggradation (sedimentation) is a second physical stressor to aquatic life evaluated in the SLRA. Criteria are not available for its evaluation, so it was assumed that aggradation would be an SOPC at any river reach where sedimentation has exceeded one meter. An aggradation threshold of one meter was conservatively used to ensure that aggradation will be evaluated further in the DLRA.

Following ANZECC (1992) guidelines, PNG (1998) recommends that DO levels in freshwater systems should be >6.0 mg/L. Similarly, the U.S. EPA (1986b) recommends a seven day average and one day minimum DO criteria of 6.0 and 5.0 mg/L, respectively, for protection of early life stages of warmwater fish.

4.3.2 Sediment

Both freshwater and estuarine sediment guidelines were identified in a manner similar to the surface water guidelines. Sources of freshwater guidelines were:

- Ingersoll et al. (1996);
- Environment Canada (1995); and
- Scientific literature.

Sources of saltwater sediment guidelines were:

- ANZECC (1997);
- Environment Canada (1995); and
- Long et al. (1995);

The individual effects thresholds for these sediment guidelines are presented in Table 19. Each guideline has at least two levels of protection associated with it. For example, the Ingersoll et al. (1996) guidelines provide sediment concentrations where there is a low likelihood of effects (Effects Range Low (ER-L) and Threshold Effect Levels (TEL)) as well as concentrations where effects are more likely to occur (Effects Range Median (ER-M) and Probable Effect Levels (PEL)). When a sediment concentration falls below ER-L and TEL values, effects are rarely observed. In contrast, the probability of effects is more frequent (generally greater than 50 percent) when concentrations exceed ER-M and PEL values (Ingersoll et al. 1996; Long et al. 1998). It should be noted that an exceedance of any one of these sediment guidelines does not necessarily mean that aquatic life are at risk. This is because the sediment guidelines are not site-specific, are conservative, and do not always indicate an effect will actually occur when exceeded (Long et al. 1998). For some metals (i.e., aluminium, chromium, copper, nickel, and zinc) certain sediment guidelines were less than background concentrations. For these cases, sediment concentrations in impacted river reaches and water bodies were compared with background concentrations. The approach for evaluating potential risks to benthos using these guidelines is provided in the Risk Characterisation (Section 5).

4.4 WILDLIFE

To evaluate a chemical's toxicity to wildlife receptors chronic toxicological effects data were obtained from the scientific literature. Chronic toxicity data are the NOAELs for the stressors evaluated. When a NOAEL was unavailable, a LOAEL was used. However, a safety factor was used to predict a NOAEL. The LOAELs and NOAELs were generally based on adverse effects to reproduction, growth, and development (Tables 20 and 21). Where chronic values are unavailable for the receptor to be evaluated, surrogate species (e.g., rat, chicken, quail) were used. In characterising effects to birds, if toxicological effects data for a bird were not available, data for mammals were used. Similarly, because little or no toxicity testing has been performed on reptiles, toxicity data on birds were used as a surrogate for reptiles. An additional safety factor of 10 was used to account for this uncertainty. The lowest most protective toxicity data were used in this assessment. In all cases, if data were available, effects on reproduction and development were evaluated over systemic and growth effects.

The potential for a chemical's toxicological effect on terrestrial invertebrates was also evaluated. Chronic toxicological effect data were obtained from the scientific literature. Toxicological data for terrestrial invertebrates are based on effects on growth, reproduction, and survival (Table 22).

Table 19. Aquatic life sediment quality guidelines (mg/kg dw).

	Ingersoll et al. (1996)				Environment Canada (1995) Freshwater		Environment Canada (1995) Saltwater		ANZECC (1997)		Long et al. (1995)		Other	
Stressor	ER-L	ER-M	TEL	PEL	TEL	PEL	TEL	PEL	ISQG-Low	ISQG-High	ER-L	ER-M	Value	Reference
Aluminium	---	58,000	---	---	---	---	---	---	---	---	---	---	---	---
Arsenic	13	50	11	48	5.9	17	7.24	41.6	20	70	8.2	70	---	---
Cadmium	0.7	3.9	0.58	3.2	0.596	3.53	0.676	4.21	1.5	10	1.2	9.6	---	---
Calcium	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Chromium	39	270	36	120	37.3	90	52.3	160	80	370	81	370	---	---
Copper	41	190	28	100	35.7	197	18.7	108	65	270	34	270	---	---
Iron	200,000	280,000	190,000	250,000	---	---	---	---	---	---	---	---	---	---
Lead	55	99	37	82	35	91.3	30.2	112	50	220	46.7	218	---	---
Manganese	730	1,700	630	1,200	---	---	---	---	---	---	---	---	---	---
Mercury	---	---	---	---	0.174	0.486	0.13	0.7	0.15	1	0.15	0.71	---	---
Molybdenum	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Nickel	24	45	20	33	18	35.9	15.9	42.8	21	52	20.9	51.6	---	---
Selenium	---	---	---	---	---	---	---	---	---	---	---	---	4	Van Derveer and Canton 1997
Silver	---	2.2[1]	---	---	---	---	0.73	1.77	1	3.7	1	3.7	---	---
Sulphate	---	---	---	---	---	---	---	---	---	---	---	---	---	---
Zinc	110	550	98	540	123	315	124	271	200	410	150	410	---	---
Mill reagents	---	---	---	---	---	---	---	---	---	---	---	---	---	---

[1]Value from Long and Morgan (1990).
TEL = Threshold effects level
PEL = Probable effects level
ER-L = Effects range-low
ER-M = Effects range-median
ISQG = Interim sediment quality guideline

Table 20. Chronic toxicity values for birds and reptiles.

Analyte	Toxicity Value (mg/kg/d)	Exposure Duration	Organism (Endpoint)	Reference
Aluminium	110	4-months	Ring dove (reproductive)	Carriere et al. 1986
Arsenic	5.14	128-days	Mallard (mortality)	USFWS 1964
Cadmium	1.45	90-days	Mallard (reproductive)	White and Finley 1978
Chromium (III+VI)	1	10-months	Black duck (reproductive)	ORNL 1996
Copper	47	10-weeks	Chicken (growth)	Mehring et al. 1960
Iron	120		Mice (teratogenicity)	U.S. EPA 1984
Lead	1.13	12-weeks	Quail (reproductive)	Edens et al. 1976
Manganese	88	224-days	Mice (reproductive)	Laskey et al. 1982
Mercury	0.412	28-days[1]	Quail (mortality)	Begearmi et al. 1980
Nickel	77.4	90-days	Mallard (growth)	Cain and Pafford 1981
Selenium	0.5	78-days	Mallard (reproductive)	Heinz et al. 1987
Silver	22.2	37-weeks	Rat (systemic)	Matuk et al. 1981
Zinc	15	44-weeks	Chicken (reproductive)	Stahl et al. 1990

[1] Considered chronic because sensitive life stage was tested (test was initiated with one day old birds).

Table 21. Chronic toxicity values for mammals.

Analyte	Toxicity Value (mg/kg/d)	Exposure Duration	Organism (Endpoint)	Reference
Aluminium	1.9	3-generations	Mouse (growth)	Ondreicka et al. 1966
Arsenic	0.126	3-generations	Rat (reproduction)	Schroeder and Mitchener 1971
Cadmium	1	6-weeks (through gestation, critical life stage)	Rat (reproductive)	ORNL 1996
Chromium (III+VI)	3.3	1-year	Rat (growth)	Mackenzie et al. 1958
Copper	11.7	357-days	Mink (reproductive)	Aulerich et al. 1982
Iron	120		Mice (teratogenicity)	U.S. EPA 1984
Lead	0.375	2-generations	Mouse (reproductive)	Schroeder and Mitchener 1971
Manganese	88	224-days	Mice (reproductive)	Laskey et al. 1982
Mercury	0.42	7-weeks[1]	Mouse (immunological)	Dieter et al. 1983
Nickel	40	3-generations	Rat (reproductive)	Ambrose et al. 1976
Selenium	0.2	2-generations	Rat (reproductive)	ORNL 1996
Silver	22.2	37-weeks	Rat (systemic)	Matuk et al. 1981
Zinc	160	16-days (through beginning of gestation, critical life stage)	Rat (reproduction)	Schlicker and Cox 1968

[1]Although not technically a chronic exposure duration, this toxicity value is low compared to other toxicity values in the scientific literature for true chronic tests (e.g., ORNL). As a result, a safety factor was not applied to the toxicity value.

Table 22. Chronic effects values for terrestrial invertebrates.

Chemical	Terrestrial Invertebrate effects value [a] (mg/kg)
Aluminium	600
Arsenic	60
Cadmium	20
Chromium	0.4
Copper	50
Iron	200
Lead	500
Manganese	100
Mercury	0.1
Nickel	90
Selenium	70
Silver	50
Zinc	10

[a] All values were derived from ORNRL 1995.

5. RISK CHARACTERISATION

The purpose of this phase of the SLRA is to integrate information on exposure and effects to provide an estimate of whether the risk exists. This estimate is presented in the form of a hazard quotient (HQ), with a quotient >1 indicating the potential for risk.

5.1 HUMAN HEALTH

5.1.1 Calculation of Health Risks for Stressors Not Causing Cancer

Human health risks are calculated by comparing estimates of exposure (lifetime chronic daily intake) with the threshold toxicity values described in Section 4.1. The calculation is shown in Equation 10 and the result is a hazard quotient (HQ).

$$HQ = \frac{\text{Chronic daily intake}}{\text{Threshold toxicity value}} \qquad (10)$$

Quotients exceeding one signify potential risks while HQs less than one suggest negligible risks. If the chemical-specific HQ exceeds one, it does not mean that there is necessarily a health risk from chronic exposure. Rather, it suggests a need for further evaluation of the chemical stressor in the DLRA.

5.1.2 Calculation of Health Risks from Cancer-Causing Stressor (Arsenic)

Arsenic is the only mine-related stressor that can potentially cause cancer. Cancer potential is estimated by multiplying the cancer slope factor[8] by an estimated lifetime average intake:

$$Cancer\,Risk = \text{Chronic daily intake}_{(lifetime)} \; x \; \text{Slope Factor} \qquad (11)$$

The cancer risk represents the probability of an individual developing cancer over a lifetime of exposure (U.S. EPA 1989a). Interpretation of cancer risk results is similar to the interpretation of non-cancer risk results in that consideration must be given to the magnitude of the risk prediction and the degree of certainty in the exposure assessment and cancer slope factor. The level of risk considered to be "acceptable" must take these factors into consideration. Specifically, the U.S. EPA assumes any exposure to arsenic is associated with a risk of cancer. This approach, however, does not take into consideration naturally occurring levels of arsenic in soil, water, and food. Such exposures often exceed U.S. EPA's lower target range[9] of 10^{-6} and may in some cases approach a

[8] The slope factor is the value used to estimate the cancer-causing potency.

[9] In the absence of other regulations, some governments (e.g., United States) have promulgated standards that consider risks in the range of one per ten thousand to one per million (10^{-4} to 10^{-6}) as protective of human health (U.S. Federal

risk of 10^{-4}. Therefore, cancer risks that exceed U.S. EPA guidelines do not necessarily indicate an actual health hazard.

As previously discussed, arsenic is known to be a human carcinogen (Section 4.1). Therefore, estimates of health risks from cancer were estimated for each river reach for which arsenic data were available. Inorganic arsenic cancer risks are expressed as unitless probabilities (e.g., $1x10^{-4}$)[10] of developing cancer during a lifetime of exposure.

5.1.3 Summary of Results for Non-Cancer Causing stressors

The results of the human health risk characterisation are summarised in Table 23 and Figures 15 and 16. The results show HQs >1 are predicted for lead and cadmium by the aquatic food consumption exposure pathway. Negligible risks are predicted for all other chemical stressors and exposure pathways.

Table 23. Summary of human health stressors of concern.

SOPC	Exposure Pathway [a]	River Reach	HQ	Reference HQ
Lead	Aquatic Food Consumption (Fish)	Estuary	1.76	< 0.01
Cadmium	Aquatic Food Consumption (Fish)	3	3.4	< 0.01
Cadmium	Aquatic Food Consumption (Shellfish)	Estuary	9.8	7

[a] The exposure pathways shown contribute the majority of the dose; and therefore, predicted risk.

Sediment data collected from the mainstem of the river near the shore (5 meters from shore) were used to estimate concentrations to which people may be exposed in Reaches 2, 4 and 5. In all cases, negligible risks were predicted from sediment exposure pathways.

Surface water exposures resulted in negligible predictions of risks for all chemical stressors and plausible pathways of exposure (i.e., skin exposure, ingesting small amounts of water over many years).

Register 53:245 1988). The World Health Organization (WHO) has based its drinking water guideline for arsenic on a 6×10^{-4} excess cancer risk (WHO 1996a).

[10] A 1 $x10^{-4}$ carcinogenic risk prediction is interpreted as a one in 10,000 "excess" incremental chance of a person developing cancer in a lifetime from exposure to the lifetime average daily intake of inorganic arsenic above any background risk. It may also be expressed as one person in 10,000 developing cancer in an exposed population. For comparison the background lifetime cancer rate in the U.S. has been estimated at between 20 to 25 percent or one in five to one in four (American Cancer Society 1993).

Figure 15. Estimated Spatial Extent of Mine-derived Sediment at Reach 3a: Upper Fly River

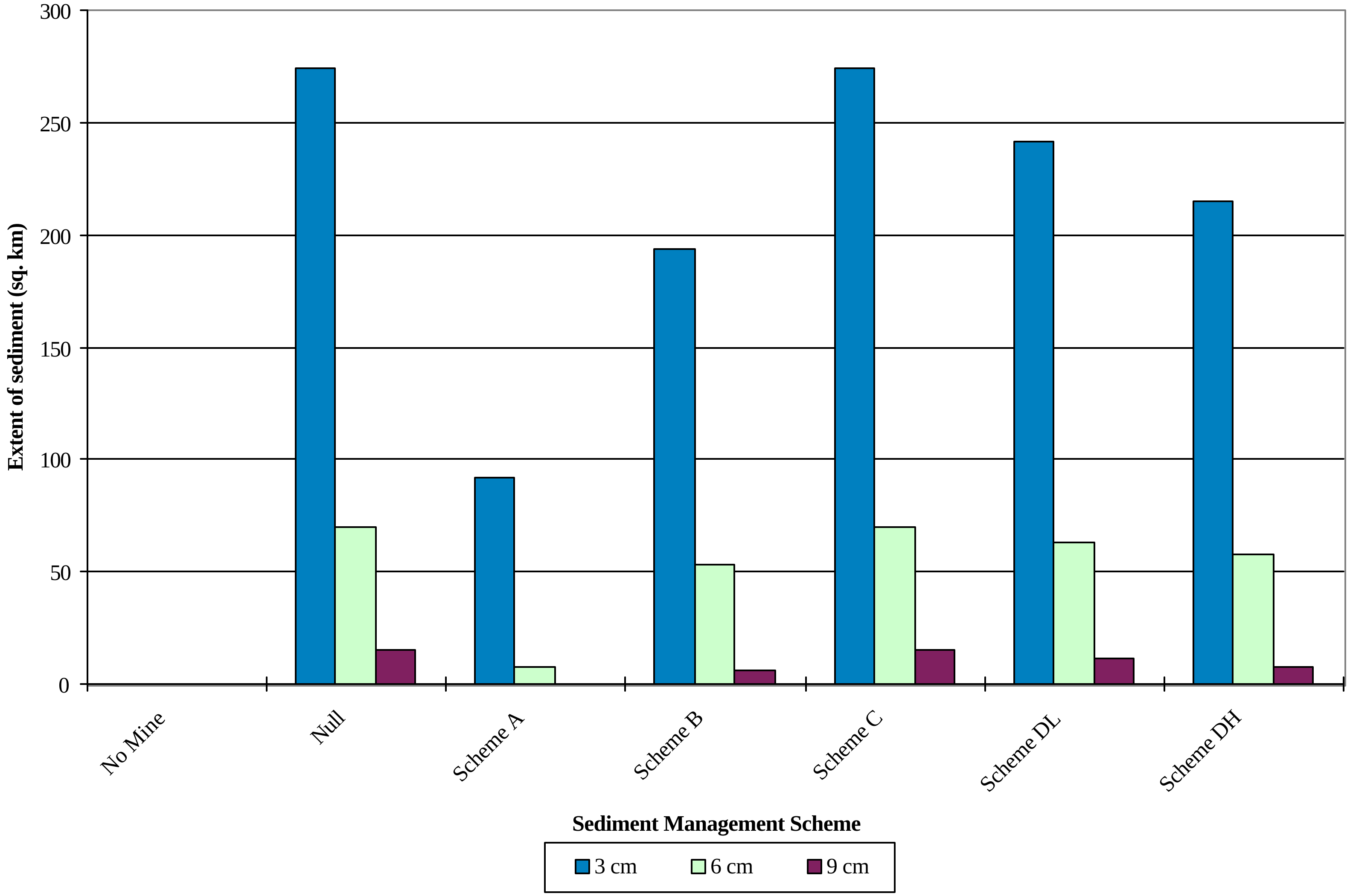
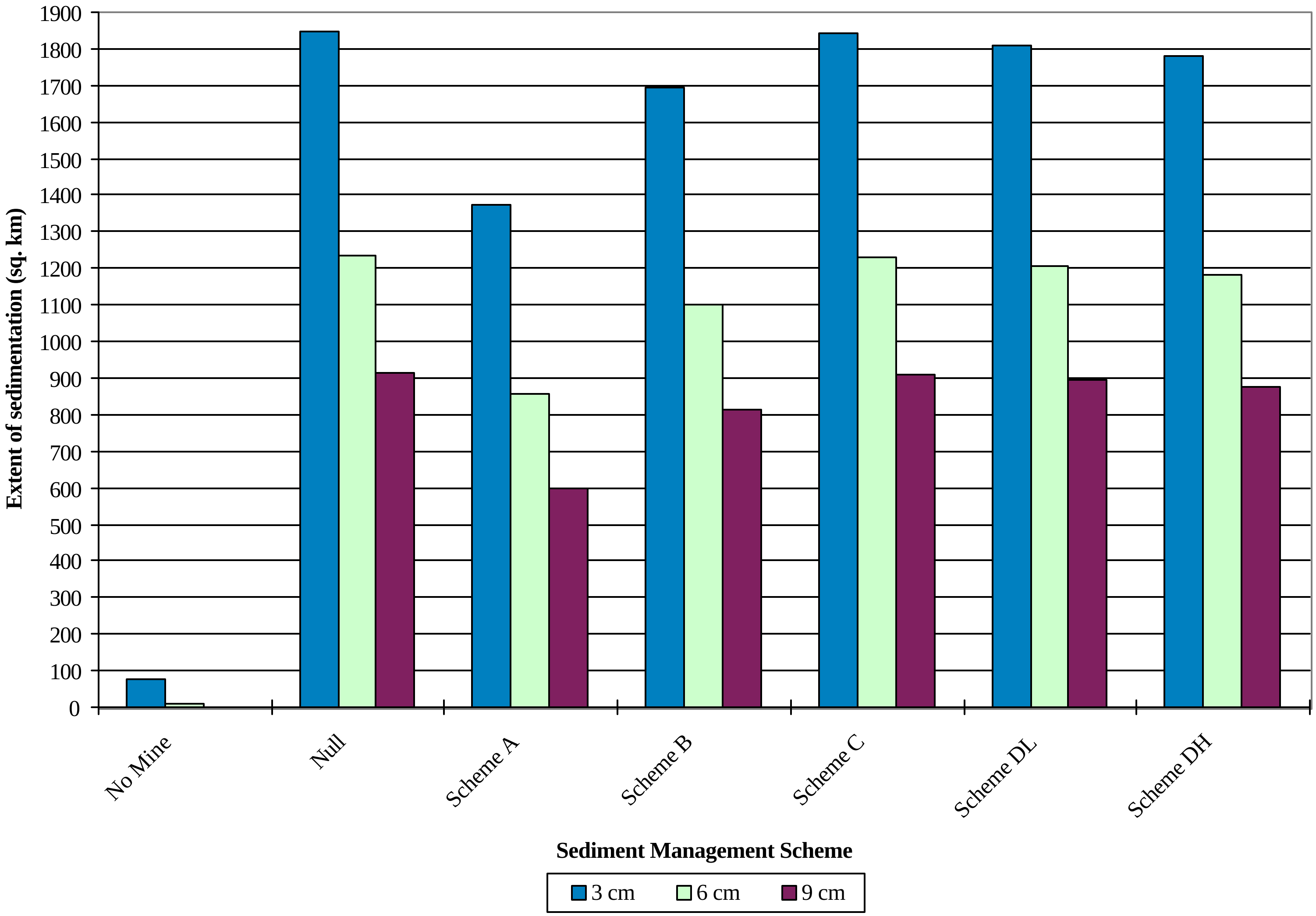


Figure 16. Estimated Spatial Extent of Mine-derived Sediment at Reach 4: Middle Fly River



Potential risks from lead exposure in the estuary are predicted from the aquatic food consumption pathway. Exceedance of the toxicity threshold in the estuary is due to consumption of mud clams that had higher chemical concentrations as compared to fish tissue. Shellfish have been reported to contain elevated metals levels (e.g., cadmium and mercury) internationally (WHO 1972). However, relative amounts of shellfish and finfish consumed by local people is not well understood. Given this uncertainty, it is possible that the relative risks from aquatic food consumption may be similar in the estuary. However, based on this exceedance, lead is retained as an SOPC in the estuary.

Potential risk from cadmium exposure was predicted in Reach 3 and the estuary based on the aquatic food consumption pathway. The cadmium criterion was also exceeded at reference sites in the estuary for aquatic foods suggesting that a portion of the predicted risk is attributable to native conditions. However, the magnitude of the exceedance was greater at the non-reference sites. As described above, this is due, in part, to elevated concentrations in shellfish.

The cadmium toxicity value is based on its accumulation in the soft tissues of the kidney and liver, it's low rate of excretion and long-half life (20-30 years). The primary source of exposure to cadmium is in foods with the highest concentrations reported to occur in liver, kidney and shellfish (WHO 1996a).

The toxicity value of 0.001 mg/kg-bw/day (intake) is set to avoid accumulation in the kidney renal cortex above 50 mg/kg. WHO has estimated that the critical concentration (in kidney) at which an adverse effect (i.e., proteinuria) would reach a prevalence of 10 percent in the exposed population is 200 mg/kg, and would require 50 years of exposure at an average intake rate of 0.175 mg/kg-bw/day (WHO 1996a; U.S. EPA 1998b). From dose-response analysis, WHO has estimated that a 10 percent prevalence of low-molecular weight proteinuria would occur after 45 years of dietary intake at 0.2 mg/day for people weighing 70 kg (JECFA 1989).

Thus, although potential risks from cadmium exposure have been predicted, many years of exposure at recently existing cadmium concentrations would be required before an effect would be observed. If cadmium concentrations were to decline after mine operations cease, then risks would decline as well. However, because potential risks have been predicted, cadmium has been retained as an SOPC for Reach 3 and the estuary.

Potential risks from the consumption of sago and cassava were evaluated. Sago were collected from the lower Ok Tedi and middle Fly River and cassava was collected in Reach 5. Therefore, the sago data was applied to Reaches 3,4, and 5 and the cassava data to Reach 5. The chemical concentrations did not appear to vary substantially between samples. The results of HQ calculations with these samples reveals that, based on the available data, no risks are predicted from long-term consumption of these foods.

5.1.4 Summary of Results for Carcinogenic Effects of Arsenic

Inorganic arsenic data were available in sediment and surface water in Reaches 2, 4, and 5. The predicted sediment cancer risks are presented in Table 24. Using the nearshore sediment data, the predicted arsenic cancer risks across all river reaches ranged from the highest predicted risk from incidental ingestion in Reach 2 of 6×10^{-5} from incidental ingestion to the lowest risk of 7×10^{-6} from skin exposure to sediment in off-river water bodies in Reach 5. The predicted risks tend to increase from Reach 5 upriver to Reach 2 generally, but with the exception of Reach 2, all risks are in the 10^{-5} range. In Reach 2, the risk reaches 10^{-4}, but this is based on the main channel sediment. For comparison, the nearshore sediment data (which are more representative of potential exposures) result in a risk prediction of 6×10^{-5}. These risks are similar to the background risk estimates that are in the 10^{-5} range.

Dissolved arsenic data were available for Reaches 2,4 and 5. Potential risks predicted using these data and assuming likely surface water exposure pathways (i.e., skin exposure and incidentally ingesting small amounts of water over many years) were very low. Only in Reach 4 did the risks predicted from incidentally ingesting surface water reach the 1 in 1,000,000 (i.e., 1×10^{-6}) level. All other surface water risks were below this level.

Table 24. Summary of cancer risks from sediment exposure.[a]

River Reach	Sediment Exposure Pathway	Predicted Cancer Risk[b]
2-Mainstem	Dermal	2×10^{-5}
2-Mainstem	Incidental Ingestion	6×10^{-5}
4-Mainstem	Dermal	2×10^{-5}
4-Mainstem	Incidental Ingestion	5×10^{-5}
5-Mainstem	Dermal	1×10^{-5}
5-Mainstem	Incidental Ingestion	2×10^{-5}
5-ORWB	Dermal	7×10^{-6}
5-ORWB	Incidental Ingestion	2×10^{-5}

[a] Based on nearshore (within 5 meters) sediment samples.
[b] No background (i.e., reference) sediment data are available.

Potential arsenic risks from the consumption of cassava were evaluated, and an excess risk of 2×10^{-4} was predicted. This result is consistent with recently conducted surveys of inorganic arsenic in food, in which the highest percentages of inorganic arsenic were found in starchy foods such as rice, yams and potatoes (Schoof et al. 1998, 1999; Slayton et al. 1996; Yost et al. 1998)[11]. These studies suggest that the diet is an important part of total arsenic intake worldwide. Based on these considerations it is likely that arsenic in cassava reflects naturally occurring conditions, although no reference cassava data are available.

[11] The majority of arsenic in foods are not in the inorganic form (Schoof et al. 1999; Yost et al. 1998). Thus, actual risks are lower than those predicted.

5.1.4.1 Field Evidence of Risk to Human Health

Flew (1998) conducted a health survey of some villages in the study area. The Flew (1998) survey evaluated such health parameters as transmission of malaria, parasite loads, malnutrition rates and sanitation. Further, the survey identified food types that are common in the diet of people living in the study area. However, this information is only qualitatively useful in the risk assessment. Information on amounts and frequency of consumption on a village-specific basis would provide a basis for refining the human health exposure assessment. Similarly, village-specific body weight data could be used to refine the exposure assessment.

Human hair data from people living along the Ok Tedi and Fly Rivers is available. Data from villages along the Ok Tedi (at various villages in Reaches 2 and 3)[12] and are summarised in Table 25. Hair data were also collected along the Fly River (collected October 19, 1998)[13] and these data are summarised in Table 26.

The reported concentrations in “normal” hair (Tables 25 and 26) are literature reported hair concentrations in unexposed people. However, no inference can be made from these data regarding any relationship between mine tailings and concentrations in hair, which may exceed these levels. Concentrations that exceed literature-based values may reflect background hair concentrations that are elevated as a result of living in a mineralogically-rich region. Thus, the apparent exceedances of literature-based values for copper, iron and manganese may reflect background hair concentrations in PNG. This hypothesis cannot be further evaluated without comparable hair data for a region of PNG that is unaffected by mine tailings.

With a few exceptions (e.g., mercury) chemical concentrations in hair are not well correlated with health effects. Moreover, the literature reported concentrations in hair are not health-based (i.e., exceedances do not imply any adverse health effects). Thus, the exceedances may be interpreted to suggest that exposures to some metals are greater for people in PNG than in other countries, but they do not suggest any potential for an adverse health effect. Because the chemical stressors in Tables 25 and 26, which exceed literature-based values (copper, iron and manganese) are homeostatically-regulated essential nutrients, health effects from these chemical stressors are unlikely (WHO 1996b; NRC 1989). If internal body concentrations exceed nutritional levels, the body will regulate these concentrations to restore them to a normal level. Thus, toxic effects will occur only when doses exceed the body’s ability to compensate for any internal concentration changes.

Copper and iron are essential components of many enzymes and other biological molecules (Matthews van Holde 1990). Iron is a component of many human enzymes (e.g., hemoglobin, myoglobin) and is stored in the human body in the spleen, liver and bone marrow (NRC 1989). Moreover, levels of body iron are regulated by changes in absorption in the intestines (NRC 1989). There are a number of enzymes in the body that contain copper and manganese.

---

[12] The villages at which this hair data set was collected are: Bongabun, Dande 2, Demsuke, Dome, Holpanai, Ieran, Kawok, Konkonda, Miamrae, Sarae, and Senamrae.

[13] The collection locations for the October 19, 1998 data set are not specified.

Based on the internal regulation and essential nutrient status of copper, iron and zinc and the absence of predicted risks using the hazard quotient method, these chemicals are not expected to have potential for adverse effects as a result of exposure to environmental concentrations.

Limited data are available on mercury concentrations in the hair of unexposed (i.e., background) people. A study on residents of Kuwait revealed concentrations in the range of 4.0 to 5.5 mg/kg (Bou-Olayan and Al-Yakoob 1994) and a study on residents of Tokyo showed concentrations from 1 to 6 mg/kg (Natkagawa 1995). These concentrations are strongly correlated with rates of fish consumption (Nakagawa 1995; Leino and Lodenius 1995). The worldwide occurrence of methylmercury in fish tissue is well known (Mason et al. 1995a,b; Matthews 1983; Marsh et al. 1995; Kyle et al. 1988; Grey et al. 1995). Thus, populations that have higher fish intake rates are expected to have higher hair mercury concentrations.

The hair mercury concentrations along the Ok Tedi and Fly Rivers tend to be higher on the lower Fly River as compared to the Ok Tedi, consistent with a higher rate of fish consumption on the lower river. Overall, the hair mercury concentrations are within the range of reported background concentrations. The maximum reported background concentration from the lower Fly River (16 mg/kg) is elevated above this range. This suggests that some individuals within the population are higher consumers of fish or have other exposures to mercury that result in higher hair mercury concentrations. Levels of mercury in hair below which no adverse effects on health are expected have been measured at 15.3 mg/kg (ATSDR 1999). However, this was the highest concentration measured in hair and no adverse effects were observed in this group (Davidson 1998). Thus, levels of mercury in hair above this concentration may not be associated with any adverse effects. Using modelling approaches the concentration in hair associated with a 10 percent increased probability of an adverse effect in a sensitive group (i.e., pregnant mothers) has been estimated at 21 mg/kg (TERA 1999). Therefore, because hair mercury concentrations are below these levels, adverse health effects from mercury exposure among the sampled populations along the Ok Tedi and lower Fly River are not expected. However, some highly exposed individuals on the lower Fly River may be approaching levels of concern.

5.1.4.2 Potential Risks to Food Resources

As discussed in the Problem Formulation, one of the assessment endpoints for the human health SLRA is potential risks to plants or animals that may be an important part of the food supply (see Sections 2.5.1 and 2.5.3.1). Reductions in the food supply could occur if populations of plants and animals declined substantially.

Table 25. Summary of metal concentrations in human hair along the Ok Tedi River.

	Hair concentrations (mg/kg)				Reported concentrations in normal hair	
Stressor	Minimum	Maximum	Arithmetic Mean	Geometric Mean	Friberg (1986)	ATSDR [a]
Arsenic	0.028	2.1	0.13	0.095	0.81	1
Cadmium	0.013	0.59	0.13	0.093	N/AV	N/AV
Copper	5.4	66	15.82	13.71	N/AV	8.9
Iron	21	830	119.32	93.63	10 – 60	N/AV
Lead	0.65	320	9.05	3.94	6 – 24 [b]	N/AV
Manganese	4.5	200	45.22	34.32	4	N/AV
Mercury[c]	0.48	3.7	1.42	1.32	N/AV	N/AV
Selenium	0.1	1.6	0.37	0.34	0.36 – 3.7	0.36 – 0.57
Zinc	60	630	173.45	157.92	N/AV	N/AV

N/AV – Not Available
[a] ATSDR- Agency for Toxic Substances and Disease Registry-Toxicological Profiles
[b] Levels in teeth and hair.
[c] Reported background mercury concentration range from 1-6 mg/kg (Bou-Olayan and Al-Yakoob 1994; Nakagawa 1995). Recent data report a no adverse effect level (NOAEL) for human hair at 15 ppm (ATSDR 1999). The level below which adverse effects are not expected has been estimated at 21 ppm (TERA 1999).

Table 26. Summary of metal concentrations in human hair along the Fly River.

	Hair concentrations (mg/kg)				Reported concentrations in normal hair	
Stressor	Minimum	Maximum	Arithmetic Mean	Geometric Mean	Friberg (1986)	ATSDR [a]
Arsenic	0.064	0.38	0.17	0.15	0.81	1
Cadmium	0.011	0.28	0.10	0.08	N/AV	N/AV
Iron	26	910	160.47	123.73	10 – 60	N/AV
Lead	0.89	200	9.32	4.18	6 – 24 [b]	N/AV
Manganese	4.5	110	30.16	23.13	4	N/AV
Mercury[c]	1.2	16	6.38	5.45	N/AV	N/AV
Selenium	0.16	0.69	0.39	0.37	0.36 – 3.7	0.36 – 0.57
Zinc	59	340	112.17	104.94	N/AV	N/AV

N/AV – Not Available
[a] ATSDR- Agency for Toxic Substances and Disease Registry-Toxicological Profiles
[b] Levels in teeth and hair.
[c] Reported background mercury concentrations range from 1-6 mg/kg. Recent data report a no adverse effect level (NOAEL) for human hair at 15 ppm (ATSDR 1999). The level below which adverse effects are not expected has been estimated at 21 ppm (TERA 1999).

Potential risks to aquatic life and wildlife (including species that may be important as food for the local people) were evaluated in the terrestrial, aquatic life and wildlife components of the SLRA. The results of that assessment revealed that, overall, chemical and physical stressors are posing risk to aquatic and terrestrial life. Specifically, reductions in fish biomass and forest dieback can be attributed to mine-related activities. On this basis, potential for adverse effects to aquatic or terrestrial resources used by local people are likely.

5.2 TERRESTRIAL VEGETATION

5.2.1 Physical Stressors

Characterisation of risk from physical stressors at this phase is designed to focus the risk assessment by screening:

1) physical stressors;
2) areas of potential risk (i.e., river reaches); and
3) receptors and endpoints (i.e., vegetation).

5.2.1.1 Summary of Risks to Terrestrial Vegetation from Physical Stressors

The basic screening approach was consistent with the HQ approach where the measured/calculated environmental concentration of a stressor is divided by an effects threshold to produce a risk quotient. For physical stressors, each appropriate parameter indicative of adverse effects for each physical stressor (e.g., sediment depth, percent annual inundation) is divided by the appropriate environmental threshold criterion to produce a risk quotient. Where this quotient is greater than 1.0 (i.e., where the environmental level for that parameter exceeds the threshold value), potential for adverse effects is indicated.

The results of the screening phase for physical stressors on terrestrial vegetation indicated that all physical stressors (floodplain aggradation, flooding, and scouring) should be retained for the DLRA; however, portions of the study area could be eliminated from further consideration. The results also indicate that some stressors pose significantly greater risk than others. Screening results are summarised below.

5.2.1.2 Areas of Potential Risk

River reaches were screened on the basis of observed adverse effects (e.g., vegetation dieback) and known or predicted exceedances of screening criteria. Hydrologic modelling results (Klohn-Crippen 1996) provided the basis for determining possible future exceedances of sediment depth and annual percent flooding. This model was used because it was the only one available at the time the SLRA was developed. Exceedances for each reach indicate that it is likely that some portion of the floodplain within each reach will have sediment accumulations that exceed the screening threshold, therefore indicating that these areas should be carried forward to be addressed in more detail in the DLRA.

Based on the threshold criteria, hydrologic modelling results, and data from the study area, Reaches 1 through 5 had exceedances for percent annual inundation and/or sediment deposition. The upper Ok Tedi (Reach 1) is largely confined to a narrow valley, but where overbank flooding occurs, there is substantial sediment deposition. Hydrologic results from Klohn-Crippen (1996) for Reaches 2-5 are presented in Figure 17.

Scouring effects were also noted for Reaches 3 and 4 (D. Reagan, personal observation; Monica Rau, personal communication). Conservative assumptions and uncertainties in the model results developed for the river system (Klohn-Crippen 1996) indicated exceedance of the 30 percent annual inundation rate for Reach 5; however, much of this reach is flood-tolerant monsoonal savannah that may not be adversely affected by flooding. Bourliere (1983b) observed that seasonal inundation for a period of seven months is a prominent feature of the savannahs of southwestern PNG, providing support for this assertion. Floodplain aggradation from mine waste disposal was predicted to exceed 30 cm in Reach 5, based on conservative assumptions.

Below Reach 5 no threshold exceedances were suggested for any physical stressors. This is due to the greatly expanded aerial extent of the floodplain that dissipates potential effects of floodplain aggradation and flooding, and the fact that terrestrial ecosystems along the lower Fly River (i.e., in Reach 6) are extremely flood tolerant. Much of the area is herbaceous swamps, but even where trees occur (e.g., Melaleuca swamps), they are tolerant of prolonged periods of inundation (Sekhran and Miller 1995).

5.2.1.3 Vegetation Types

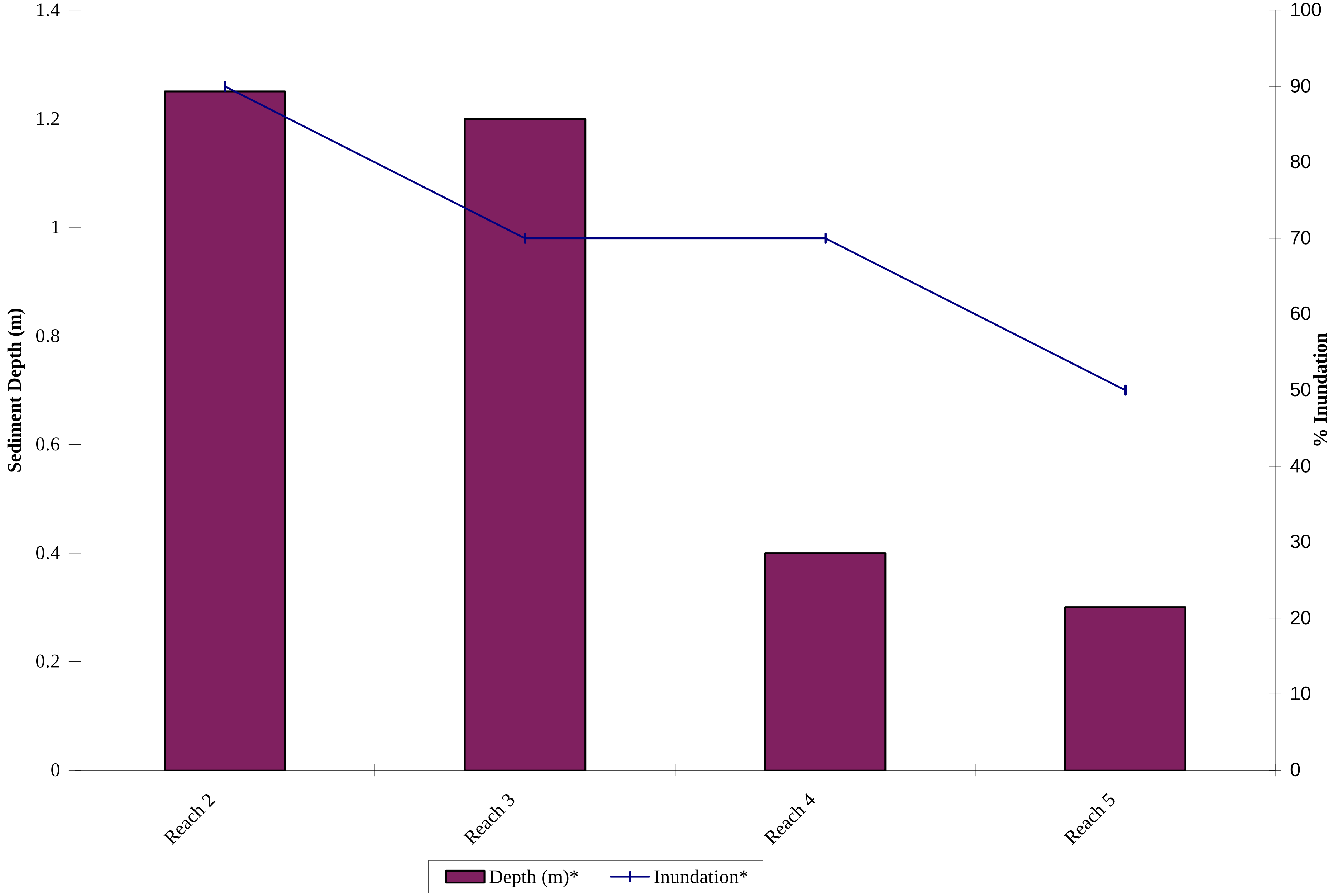
All terrestrial vegetation types present within Reaches 1-5 are retained as receptors, based primarily on the potential for adverse effects on all vegetation types from aggradation. Flooding effects are known for forest types; however, open savannahs on the floodplain in Reaches 4 and 5 are flood-tolerant and are not expected to experience as severe adverse effects of increased inundation as occur in forested areas. In general, woody species are more susceptible to prolonged flooding than are grasses and sedges that dominate these savannahs.

5.2.1.4 Field Evidence of Risk to Terrestrial Vegetation

Floodplain aggradation, flooding, and scouring are natural processes that occur in all river systems. Identification of these factors as mine-related physical stressors is based on the incremental increase in these factors caused by mine waste disposal practices. Evidence of adverse effects due to the increased intensity and/or duration of the physical stressors is provided in recent investigations performed by OTML that document increased sedimentation rates in shallow flooded areas, including floodplains, that support terrestrial vegetation (OTML 1998b). Examination of aerial photography combined with hydrologic monitoring have documented increased depth and duration of flooding in portions of the Ok Tedi and Fly River basins as a result of the increased sedimentation.

Duff (1992) documented four symptoms of vegetation stress within the study area including: extensive leaf loss, tree death, leaf chlorosis, and changes to understory microclimate and light

Figure 17. Floodplain Sediment Depths and Annual Percent Inundation



levels due to direct physical damage and burial by sediment. These effects appeared to be the result of various combinations of increased floodplain aggradation and flooding frequency but not from possible chemical contamination from mine waste disposal. Prendergast et al. (1996) document root zone water logging, physical damage, and sediment deposition as effects of physical stressors on terrestrial vegetation.

Evidence of scouring across meanders is apparent in recent aerial photos and from direct observation (D. Reagan, personal observation; Monica Rau, personal communication). This scouring appears to be the direct result of increased flood levels that increase the flows across these low areas (Klohn-Crippen 1996).

5.2.2 <u>Chemical Stressors</u>

The following provides a discussion of potential risks to terrestrial vegetation communities based on a comparison of whole-sediment analytical data from each river reach with phytotoxicity-based thresholds discussed in Section 3.2

Figures 18 through 26 present the HQs for SOPCs with the potential to pose an unacceptable risk to terrestrial vegetation communities within the Ok Tedi/Fly River system. Appendix B includes a complete summary of sediment analytical data and HQ values.

5.2.2.1 Reaches 1 and 2: Mid-channel Sediment Quality

The mine-derived sediments along the riverbank in Reaches 1 and 2 may potentially pose an unacceptable risk to riparian vegetation communities exposed to the sediments. Reaches 1 and 2 are, in general, not floodplain depositional areas, and terrestrial vegetation communities away from the riverbank are unlikely to be exposed to riverine sediments.

A total of eight SOPCs have been identified in riverbank sediments in Reach 2 at concentrations in excess of the respective phytotoxicity-based soil threshold concentrations and estimated natural background sediment concentrations from Reach 4. These SOPCs and their respective HQs are:

- Arsenic (HQ 4.4);
- Copper (HQ 29);
- Iron (HQ 2.1);
- Manganese (HQ 1.9);
- Molybdenum (HQ 9.4);
- Lead (HQ 7.8);
- Silver (HQ 1.3); and
- Zinc (HQ 2.5).

5.2.2.2 Reach 3/3a and Reach 4 Sediment Quality

No sediment data were available from Reach 3 (lower Ok Tedi River) or Reach 3a (Fly River above Ok Tedi). However, sediment data from Reaches 2 and 4 indicate that potential for sediments in

Reach 3 and 3a to contain of several SOPCs, including copper, molybdenum, lead, arsenic, zinc, iron, manganese and silver, at concentrations above the phytotoxicity-based threshold values. Of these, SOPCs with the potential to pose an unacceptable risk to floodplain vegetation communities include copper, molybdenum, lead, and silver.

UNEP (1995) indicated that there was a distinct signature between underlying natural and overlying mine-related sediments on the floodplain in Reach 4. Mine-related sediments in Reach 4 showed significant enrichment for some SOPCs, while some occurred at lower concentrations. In Reach 4, molybdenum had the highest enrichment factor (factor 23), followed by copper (factor 12), lead (factor 4.4), calcium (factor 4.1), silver (factor 3.5), sulphur (factor 2.6) strontium (factor 2.5), and zinc (factor of 1.6).

5.2.2.3 Reach 5 Sediment Quality

In Reach 5 (lower middle Fly River), SOPCs with the potential to pose an unacceptable risk to terrestrial vegetation communities include copper, lead, molybdenum, and silver.

The sediment model of Parker and Cui has predicted negligible (<3 cm) accumulation of mine-related sediments on the floodplain in Reach 5 within the next 70 years. However, existing floodplain sediment data from Reach 5 indicate that elevated concentrations, with respect to natural background sediment chemistry and phytotoxicity-based guidelines, of several analytes already occur in the floodplain sediments. The source of the SOPCs has not been established definitively; however, the suite of analytes, upriver depositional regime, and lack of other known sources of contamination would indicate that the floodplain sediments in Reach 5 are mine-related. The current extent of mine-derived sediments in Reach 5 or other reaches has not been definitively identified or mapped. Such information would verify whether the Parker and Cui model prediction has underestimated the potential floodplain sediment deposition in Reach 5 and other reaches.

5.2.2.4 Reach 6 (lower Fly River and Estuary) Sediment Quality

No floodplain sediment data were available from Reach 6. The data from Reach 5 may be applicable for Reach 6; however, SOPC concentrations are likely to be less than those measured in Reach 5. Sediment data from Reach 6 are required before a definitive statement of risk can be made. Mid-channel estuarine sediment data indicate that the concentrations of SOPCs for which data are available are below phytotoxicity-based threshold levels.

Figure 18. Arsenic Hazard Quotients for Terrestrial Vegetation.

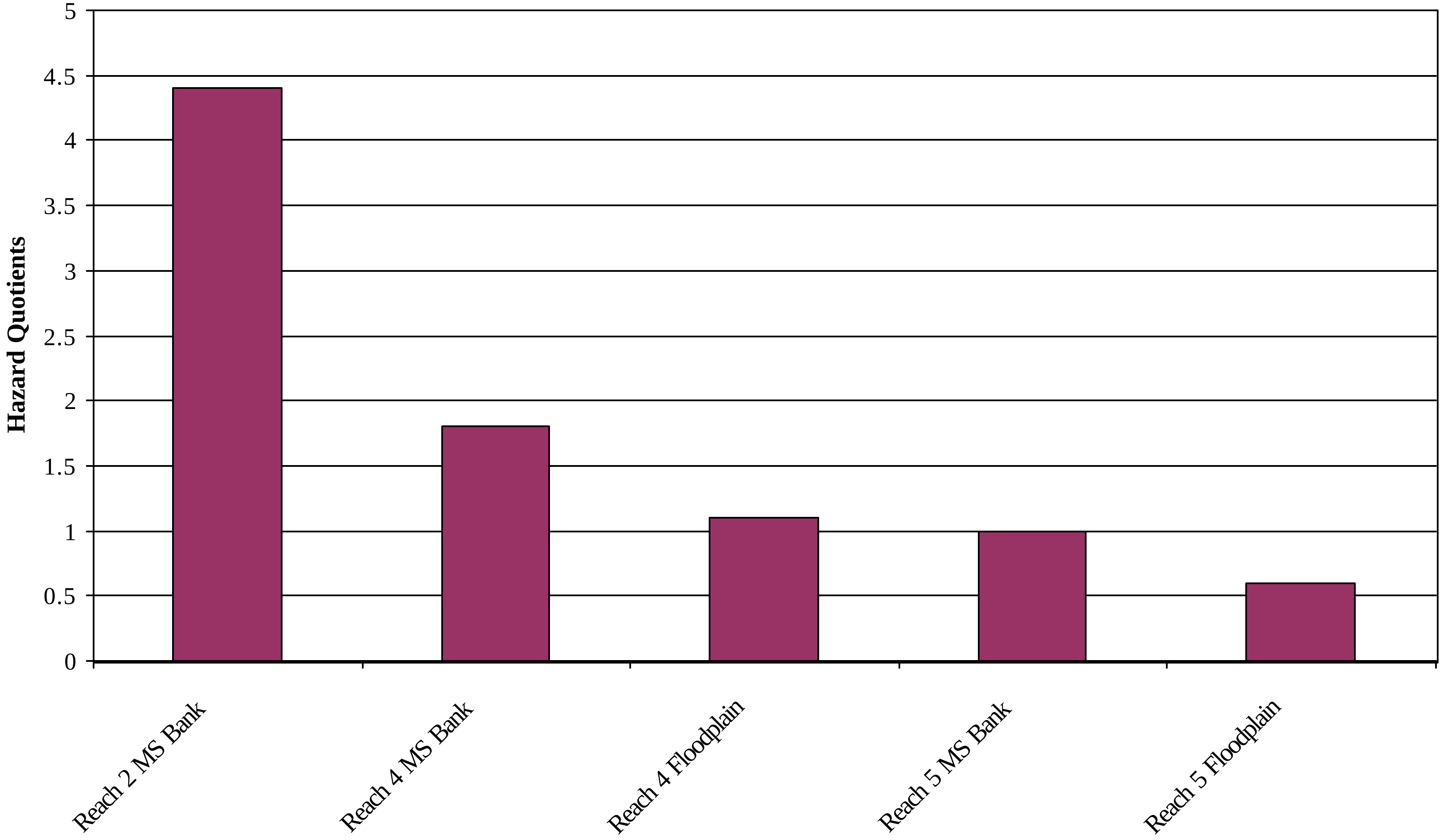


Figure 19. Cadmium Hazard Quotients for Terrestrial Vegetation.

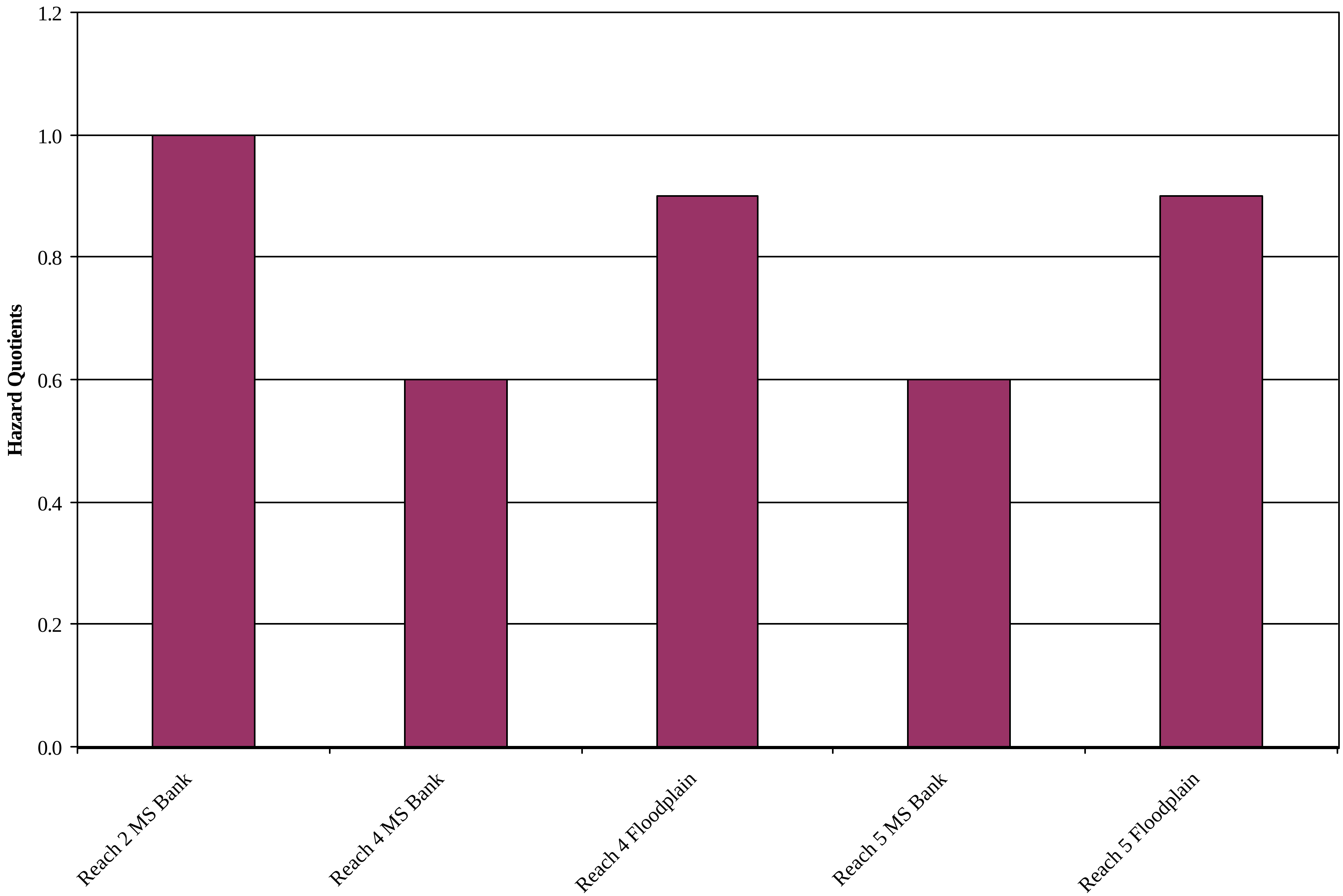


Figure 20. Copper Hazard Quotients for Terrestrial Vegetation.

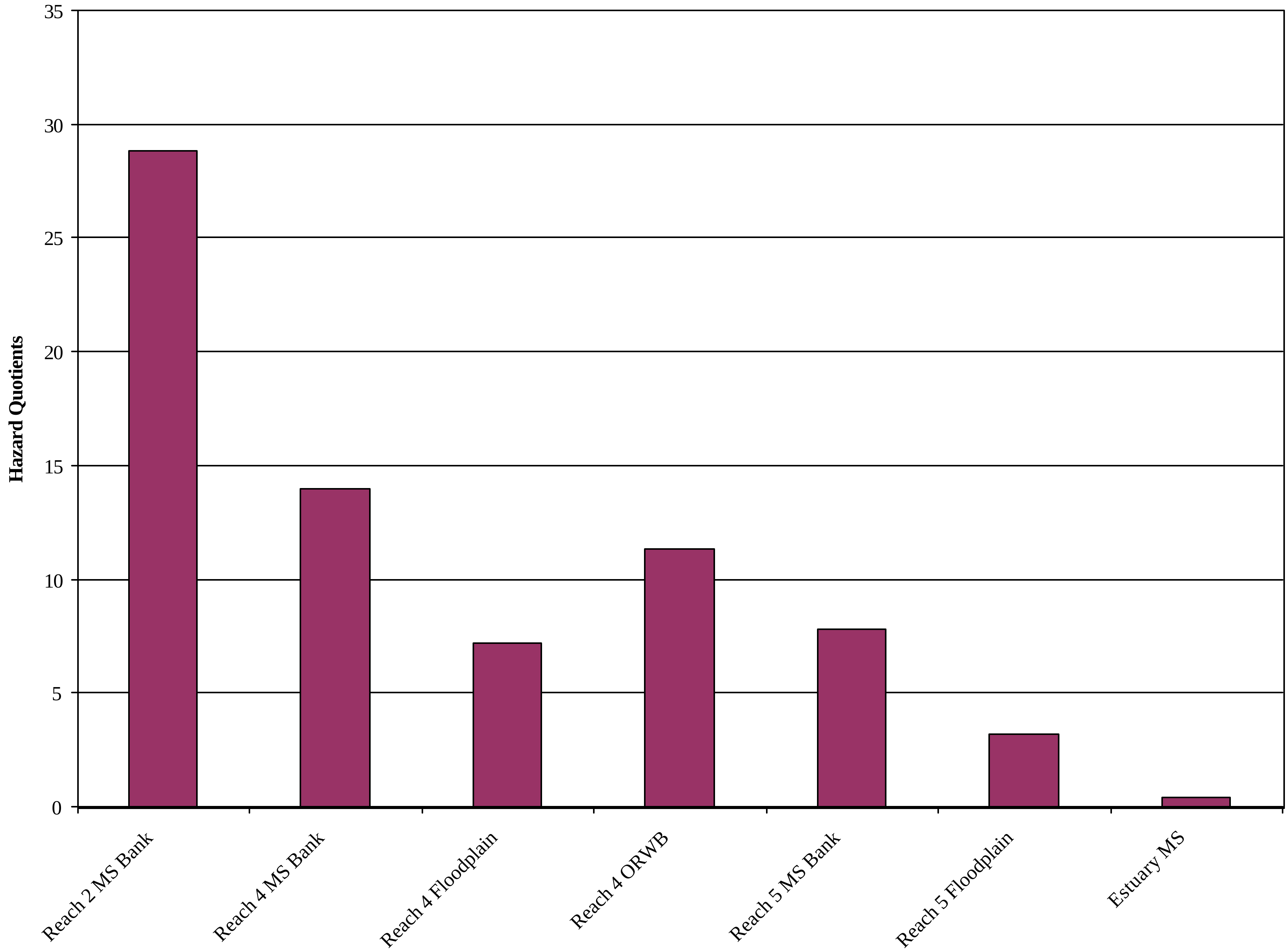


Figure 21. Iron Hazard Quotients for Terrestrial Vegetation.

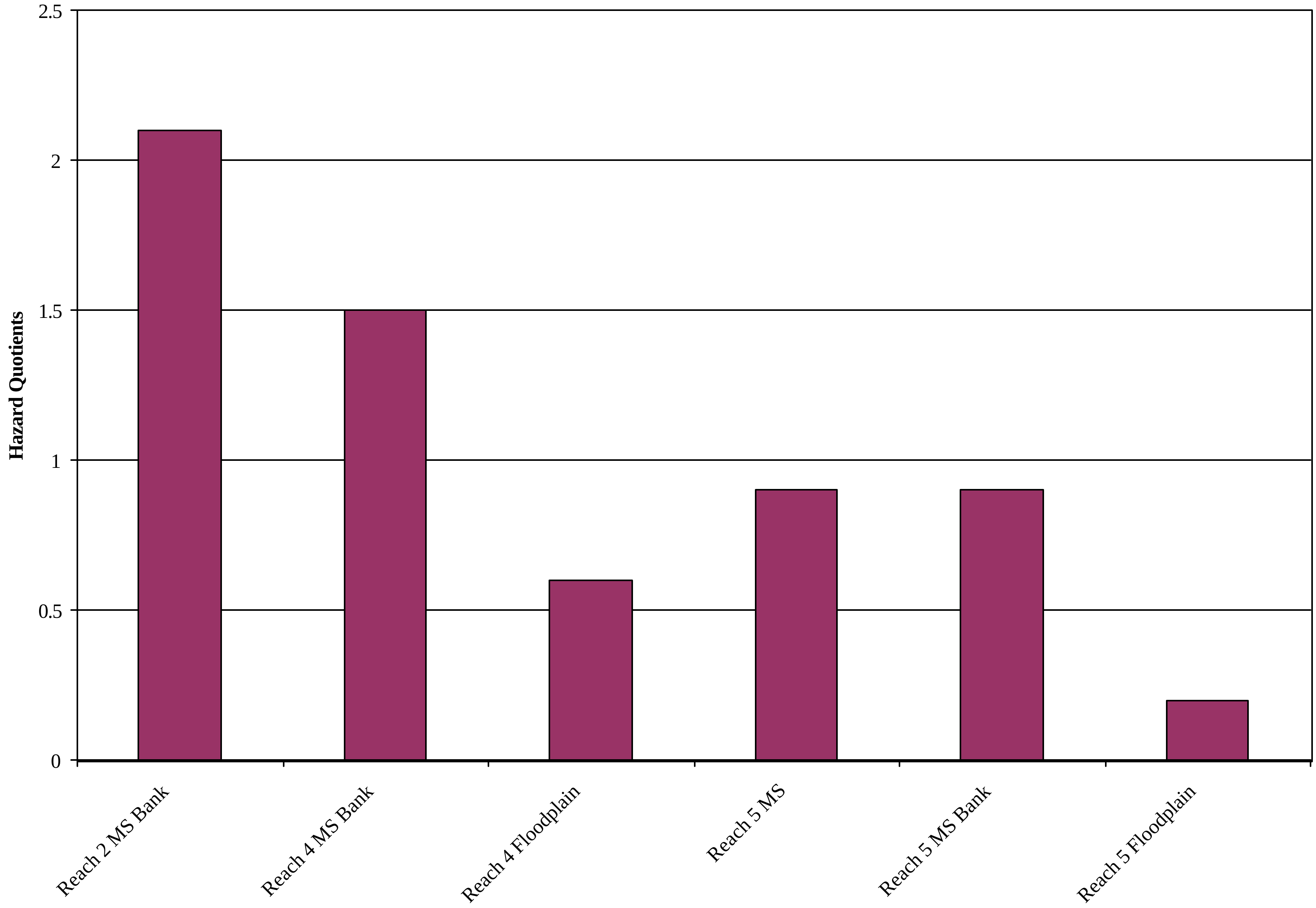


Figure 22. Lead Hazard Quotients for Terrestrial Vegetation.

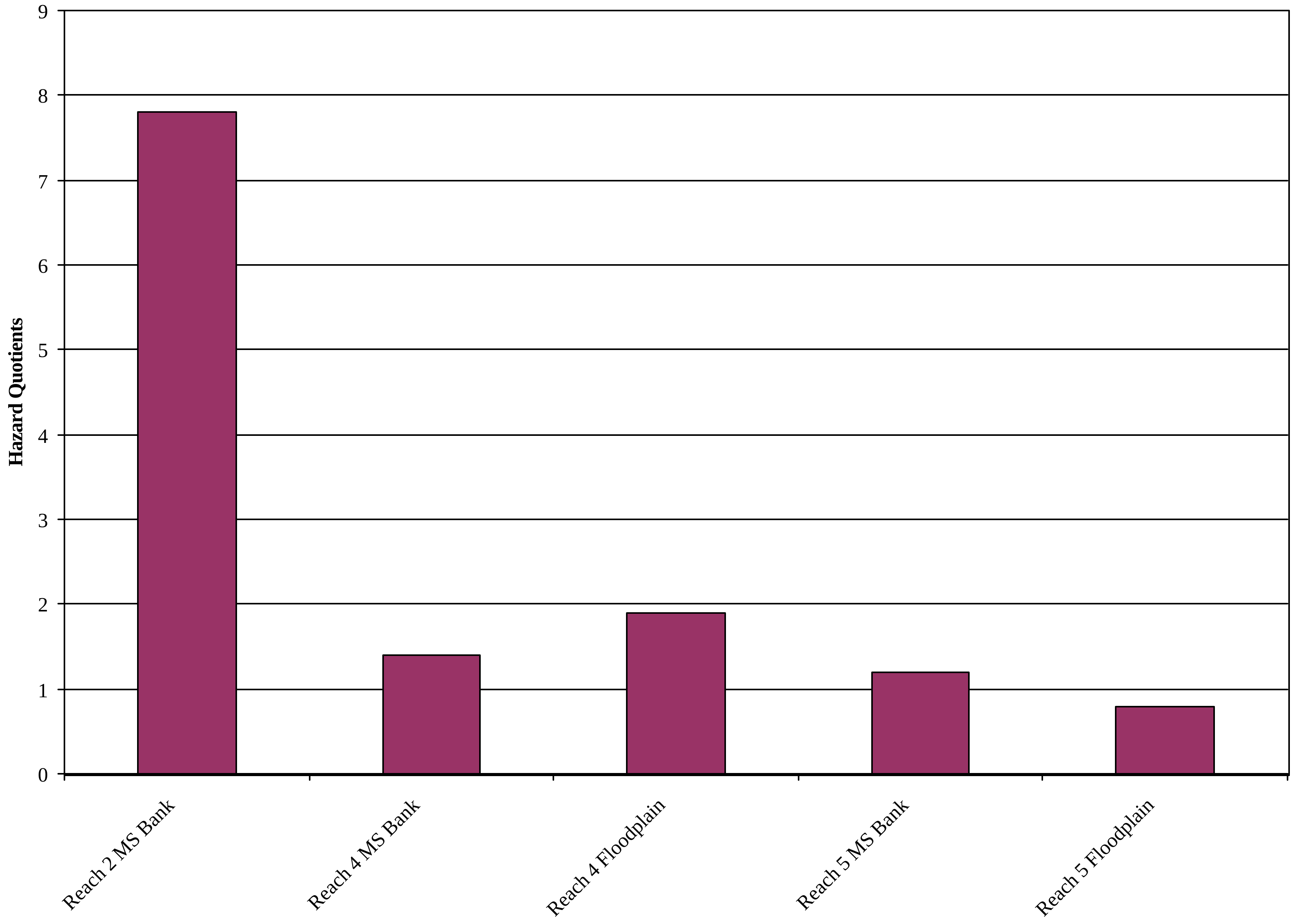


Figure 23. Manganese Hazard Quotients for Terrestrial Vegetation.

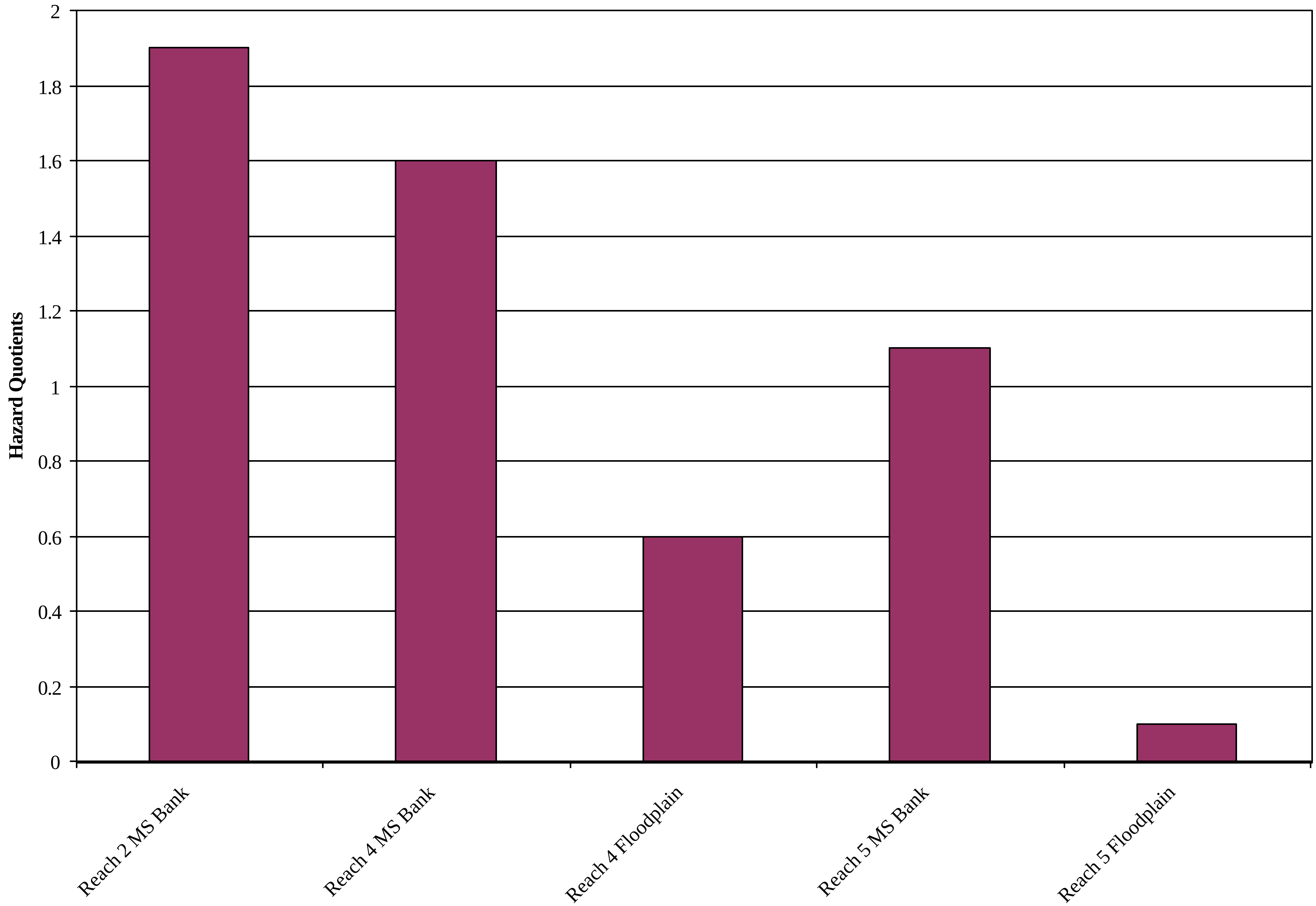


Figure 24. Molybdenum Hazard Quotients for Terrestrial Vegetation.

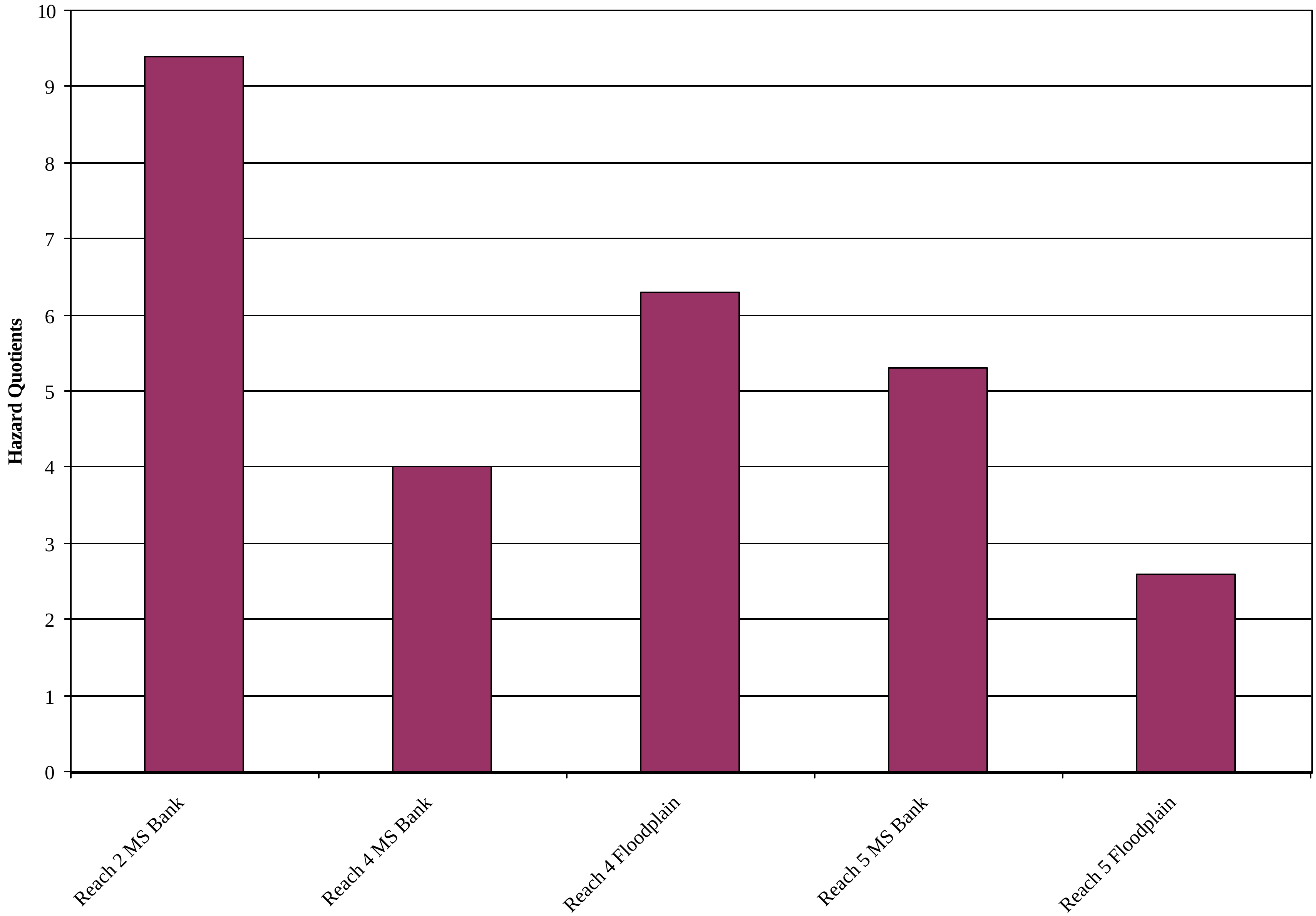


Figure 25. Silver Hazard Quotients for Terrestrial Vegetation.

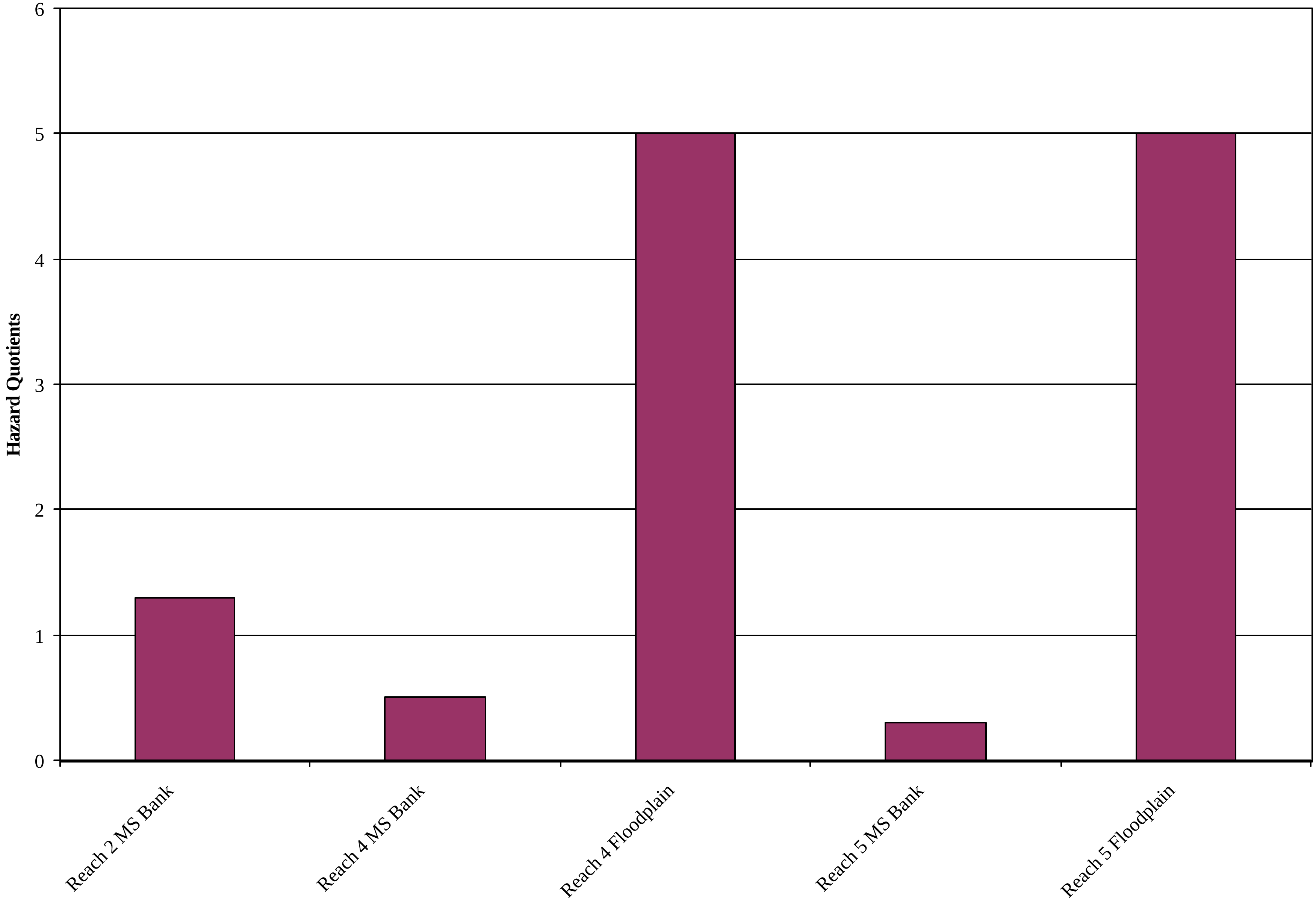
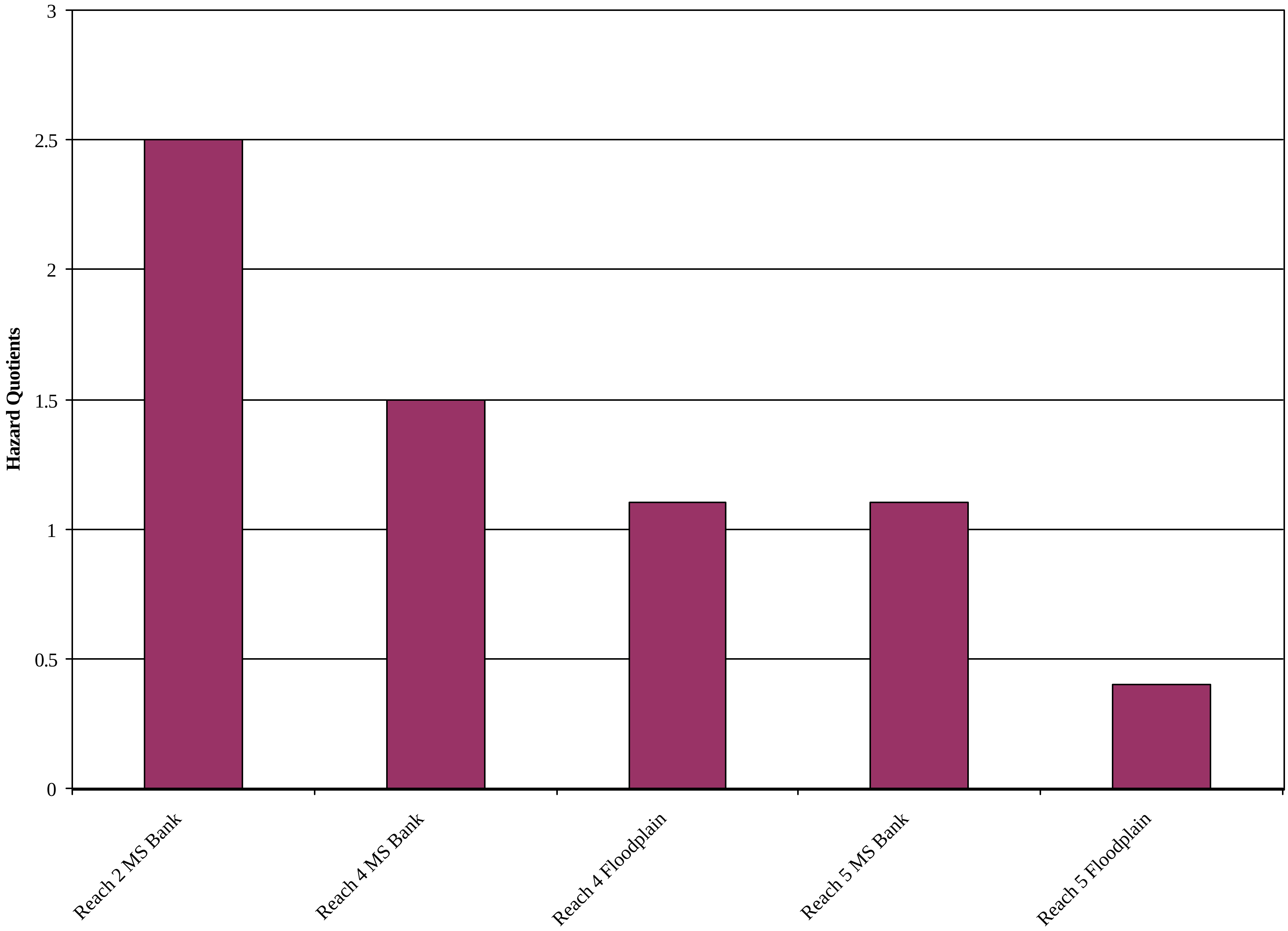


Figure 26. Zinc Hazard Quotients for Terrestrial Vegetation.



SOPCs Concentration Changes with River Distance

As indicated in Figure 27, the concentration of copper in sediments is higher nearer the mine, and the concentration declines with distance away from the mine. This trend is also evident for other SOPCs (e.g., lead, molybdenum, and zinc). Copper concentrations are similar in riverbank sediments and ORWBs in Reach 4, and potentially other reaches. Copper concentrations in floodplain sediments are lower than copper in riverbank sediments; and generally the data for other SOPCs indicate similar concentration gradients. Exceptions to this include lead, molybdenum and silver, which apparently occur at higher concentrations on the floodplain than in riverbank sediments.

UNEP (1995) indicated that the concentration of SOPCs in sediments decline with distance down river mainly due to two processes.

- Sediment admixture/erosion; and
- Mobilisation of metals from the solid phase.

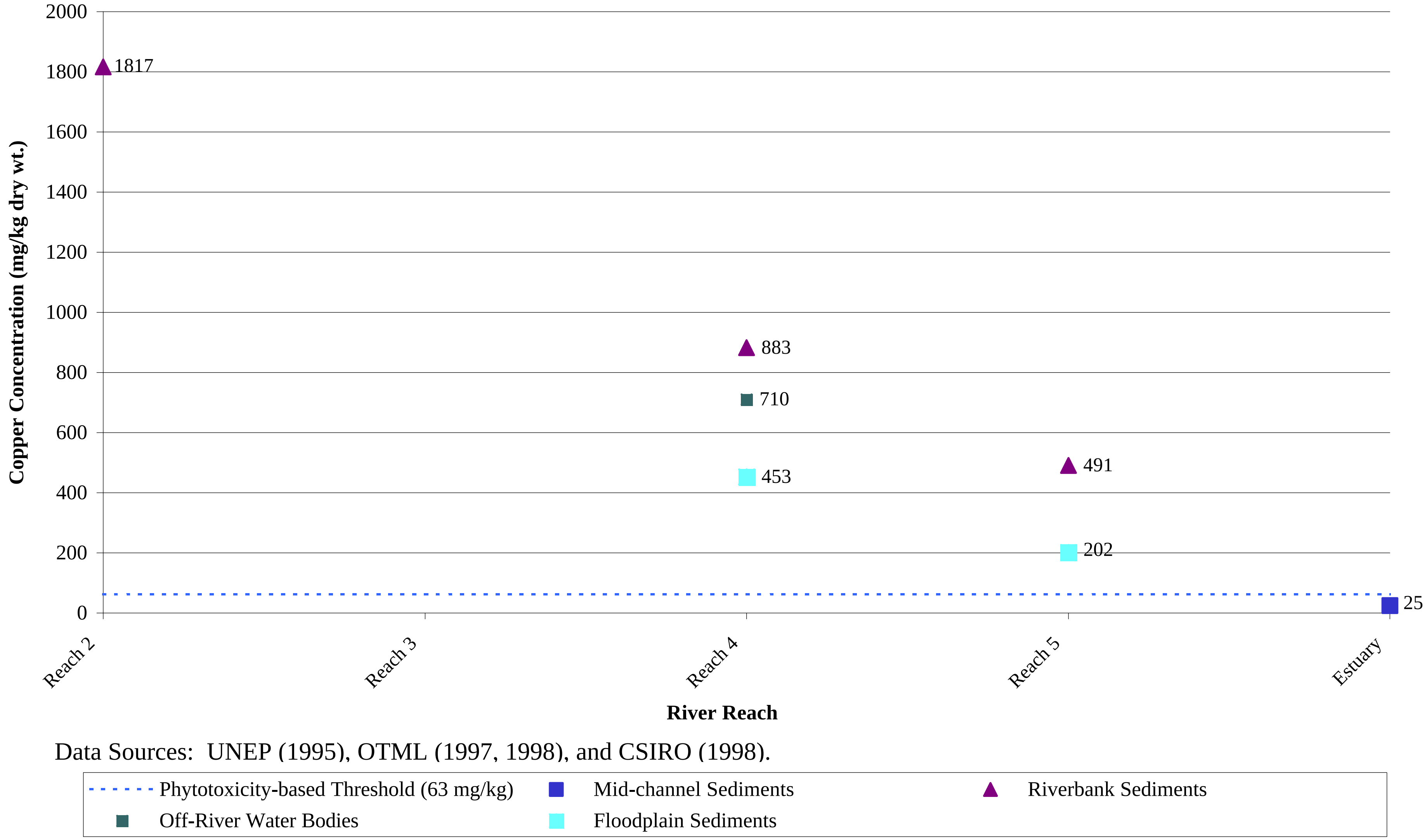
The upper Ok Tedi River receives water and suspended sediments from several tributaries. Further dilution in SOPC concentration occurs below the confluence of the Ok Tedi and Fly Rivers. However, as the Fly River and other tributaries have a much lower suspended load than the Ok Tedi River, the dilution effects are small. Lateral erosion in the winding section of the middle Fly River (Reach 4) may be an important process. Admixture (blending) of weathered floodplain sediments in the river course is responsible for increasing concentrations of several analytes (i.e., zirconium, scandium, titanium, and chromium) in the mine-related sediments in the middle Fly River floodplain (UNEP 1995), and presumably for reducing the sediment concentration of mine-related analytes (e.g., copper, lead, molybdenum, zinc).

Mineral dissolution occurs during particulate transportation down river. Mine-derived metals associated with calcite and sulphides (e.g., pyrite and chalcopyrite) may go into solution, at least partially; if dissolved, the metals may either remain in solution or become adsorbed onto particulate matter (e.g., iron [III] oxyhydroxides or organic matter).

5.3 AQUATIC LIFE

The methods and results of the aquatic life risk characterisation are presented in this section. In addition, evidence of risk from field surveys is discussed in relation to the HQ results.

Figure 27. Copper Concentrations in Whole-Sediments: Ok Tedi/Fly River Systems.



Data Sources: UNEP (1995), OTML (1997, 1998), and CSIRO (1998).

5.3.1 Risk Characterisation Methods and Results

Chemical stressors and TSS were screened in the SLRA using the HQ approach (Barnthouse et al. 1986; Suter et al. 1992), where:

$$HQ = \frac{\text{Expected Environmental Concentration}}{\text{Criterion}} \qquad (12)$$

For surface water, an HQ > 1 suggests that the stressor may be present at a high enough concentration to adversely affect aquatic communities. Note that an HQ > 1 does not mean that a stressor is adversely affecting the aquatic community, only that it may potentially be affecting the community and should be evaluated further in the DLRA. All HQ calculations can be found in Appendix C.

For sediment, a weight-of-evidence approach was used to identify SOPCs in sediment. A variety of different types of sediment guidelines (e.g., ER-L, ER-M) from multiple sources (e.g., Long et al. 1995; Environment Canada 1995) were identified in the effects characterisation (Section 4). The ability of these different types of guidelines to predict toxicity (or lack of toxicity) to benthic organisms was reviewed by Long et al. (1998). Long et al. assessed the toxicity of hundreds of field-collected sediment samples using various laboratory bioassays. Based on the data provided in their paper, it is clearly evident that several ER-L values, and even more TEL values, need to be exceeded before sediment toxicity is observed with any consistency. For example, based on the amphipod survival test, the percentage of non-toxic samples when no ER-L values were exceeded was 68 percent, while the percentage ranged from 50 percent to 89 percent (mean of 63.5 percent) when one to ten ER-L values were exceeded.

To further evaluate the data, we conducted a step-wise ANOVA to determine how many ER-L values need to be exceeded before excess sediment toxicity is observed (i.e., before the degree of toxicity is significantly different than when no ER-L values are exceeded) and how many TEL values need to be exceeded before excess sediment toxicity is observed. The results of these analyses demonstrated that four or more ER-L values and nine or more TEL values need to be exceeded before the percentage of toxic samples is significantly greater than the percentage of toxic samples when zero ER-L values or zero TEL values are exceeded, respectively. Accordingly, the following rules were developed for identifying sediment SOPCs in the SLRA:

1) If a metal exceeds its ER-M or PEL value, it is an SOPC; and
2) If four or more metals exceed their ER-L values for a given site, or nine or more metals exceed their TEL, those metals are considered SOPCs.

These conditions are set for SOPC selection because the sediment guidelines are not site-specific, are conservative, and do not always indicate an effect will actually occur when exceeded (Long et al. 1998). Furthermore, mixtures of toxic substances in the Ok Tedi/Fly River System are expected to be less complex chemically than the data upon which the sediment guidelines are based. The guidelines are based on sediments containing mixtures of metals, polychlorinated biphenyls, polycyclic aromatic hydrocarbons, and other organic chemicals from a variety of industrial sources.

The guidelines do not differentiate toxicity by chemical source. Because these sediment guidelines are not site-specific and cannot definitively identify SOPCs, an additional analysis was undertaken to evaluate sediment risks.

In surface water, cadmium, copper, iron, lead, selenium, and TSS HQs exceeded one in various river reaches. Copper had HQs exceeding one as far downstream as Reach 5 while TSS HQs exceeded one as far downstream as Reach 6 (Table 28 and Figures 28 through 32). Copper had HQs exceeding one for acute effects but not chronic effects in Reaches 3 through 5. This is a function of the apparent copper spikes in the system that are considerably higher than the long-term average (i.e., chronic) concentrations. Figure 28 shows the 95% UCL of the mean concentration of copper in sediments in river sediment. The data indicate that copper concentrations are higher nearer the mine, and the concentration declines with distance away from the mine. TSS HQs were greater than one in every reach evaluated (Figures 33 through 35). Although TSS is undoubtedly a stressor of potential concern in the Fly River, the magnitude of the HQs may be questionable because it is unknown how relevant the literature-based criteria are to the Fly River system. Uncertainties associated with the surface water HQs are described in Section 5.5.

As discussed in previous sections, DO levels have naturally high variability in many water bodies within the Ok Tedi/Fly River system, particularly in ORWBs. In Reaches 1, 2, and 3, the 95% lower confidence limits (95LCL) on the mean DO concentrations were all greater than 6 mg/L, the criterion recommended by PNG (1998) and ANZECC (1992). In ORWBs in Reaches 4 and 5, 95LCL (mean) DO concentrations were 4.7 and 3.5 mg/L, respectively. For comparison, the 95LCL (mean) DO concentrations in reference ORWBs at Kiunga were 5.4 mg/L. Dissolved oxygen levels appear to be the lowest on the floodplain. In Reach 5, for example, the 95LCL (mean) DO concentration was 1.7 mg/L. Since DO levels on the floodplain can be naturally low, and due to the lack of DO data for the floodplain at a reference site, it is not known if mining operations are exacerbating hypoxia on the floodplain. Continued monitoring of DO levels, particularly in ORWBs and on the floodplain, is recommended.

In sediments, the metals with HQs exceeding one are shown in Table 29. In addition, metals with a maximum HQ of at least 0.5 are shown graphically in Figures 36 through 42 (all HQs are tabulated in Appendix C). The HQs for most metals are highest in Reach 2 and then decrease in downstream reaches. Aggradation is also considered a SOPC for Reaches 2 through 5 because mean annual elevation changes due to aggradation are greater than one meter (OTML 1997a). Uncertainties in these HQs are described in Section 5.5.

The reasons for these declines, however, are not clear. It is possible that changes in ORWB communities may be linked to changes in river levels, but this relationship is not clear. In the estuary, no significant between-estuary differences in the number of fish species, number of fish, and fish biomass were detected, indicating there have been no detectable mine-derived effects on fish in the Fly River estuary (OTML 1998a). It should be noted, however, that the community composition could change in response to mine-related effects without a significant impact on the number of species, number of fish, or biomass (OTML 1998a). In addition, the fish catches had very high within and between site/sample variance (OTML 1998a) which would make detection of significant differences in between-estuary fish catches difficult.

Table 28. SOPCs in surface water.

Location	Site-Type	SOPCs	
		Acute	Chronic
Reach 1	Main Stem	None	TSS
	Main Stem Reference	None	None
Reach 2	Main Stem	Cu, TSS	Cu, Se, TSS
	Main Stem Reference	---	---
Reach 3	Main Stem	Cu, TSS	Cd, TSS
	Main Stem Reference	None	TSS
	ORWB	---	---
	ORWB Reference	None	None
	Floodplain	---	---
	Floodplain Reference	---	---
Reach 4	Main Stem	Cu, TSS	TSS
	Main Stem Reference	---	---
	ORWB	Cu	---
	ORWB Reference	---	---
	Floodplain	---	---
	Floodplain Reference	---	---
Reach 5	Main Stem	Cu, TSS	Fe, Pb, TSS
	Main Stem Reference	TSS	TSS
	ORWB	None	Fe, TSS
	ORWB Reference	---	---
	Floodplain	Cu	Fe, TSS
	Floodplain Reference	---	---
Reach 6	Main Stem	TSS	TSS
	Main Stem Reference	---	---
	ORWB	---	---
	ORWB Reference	---	---
	Floodplain	---	---
	Floodplain Reference	---	---
Estuary		---	---

--- = No data

Table 29. SOPCs in sediment.

Location	Site-Type	SOPCs
Reach 1	Main Stem	---
	Main Stem Reference	---
Reach 2	Main Stem	Ag, As, Cd, Cu, Pb, Mn, Zn
	Main Stem Reference	---
Reach 3	Main Stem	---
	Main Stem Reference	---
	ORWB	---
	ORWB Reference	---

Table 29. SOPCs in sediment (continued).

Location	Site-Type	SOPCs
	Floodplain	---
	Floodplain Reference	---
Reach 4	Main Stem	As, Cd, Cu, Pb, Mn, Zn
	Main Stem Reference	---
	ORWB	Cu
	ORWB Reference	---
	Floodplain	Ag, Cu, Pb, Zn
	Floodplain Reference	---
Reach 5	Main Stem	Cd, Cu, Pb, Zn
	Main Stem Reference	---
	ORWB	Al, Cu, Pb, Zn*
	ORWB Reference	---
	Floodplain	Cu
	Floodplain Reference	---
Reach 6	Main Stem	None
	Main Stem Reference	---
	ORWB	---
	ORWB Reference	---
	Floodplain	---
	Floodplain Reference	---
Estuary		---

--- = No data
* Although only three metals (Cu, Pb, Zn) exceeded the ER-L, aluminium exceeded the ER-M and therefore theoretically also would have exceeded the ER-L (if it was available). Therefore, lead and zinc were included as if all four metals (Al, Cu, Pb, Zn) exceeded the ER-L.

5.3.2 Field Evidence of Risk to Aquatic Life

Biological monitoring of the Fly River and its associated floodplain and ORWBs has revealed decreasing fish biomass in many of the river reaches. For example, fish catches at sites along the Fly River at Kuambit in Reach 4 have been decreasing since 1994 (OTML 1998a). At Obo in Reach 5 and Ogwa in Reach 6, fish catches had also shown a decreasing trend, but they increased in 1997. At riverine sites, the greatest long-term reduction in biomass (89.3 percent decrease in gill nets and 100 percent decrease in seine nets) has been observed in the Ok Tedi at Ningerum (Reach 2) (OTML 1998a). The next greatest reduction was observed at Bosset (Reach 5, 75.6 percent). Since the commencement of biological monitoring at various ORWB sites, fish catches have fluctuated, although the data indicate that species diversity and fish biomass have declined (OTML 1997a).

Figure 28. Copper Hazard Quotients for Aquatic Life: Acute Surface Water.

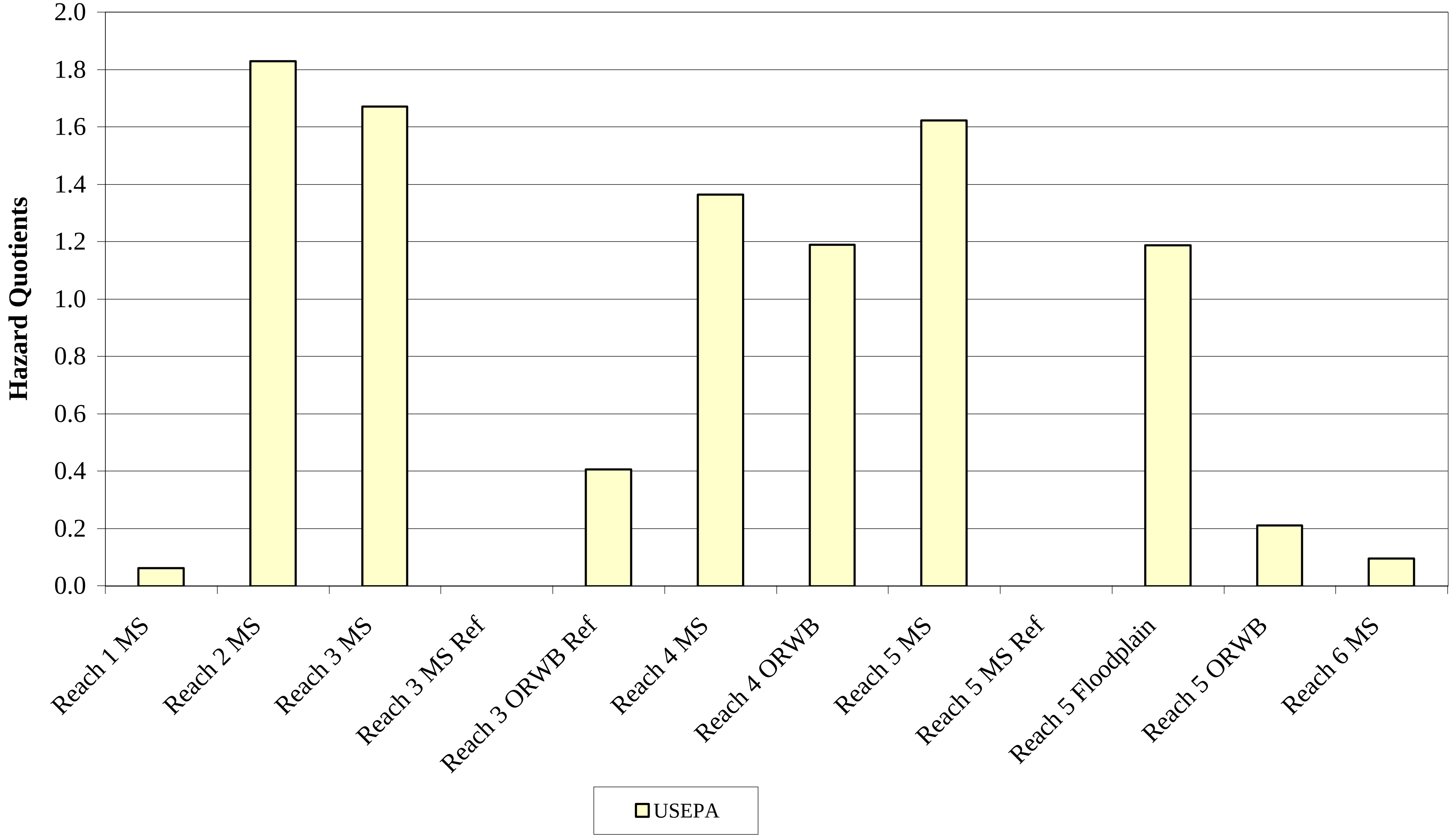


Figure 29. Cadmium Hazard Quotients for Aquatic Life: Chronic Surface Water.

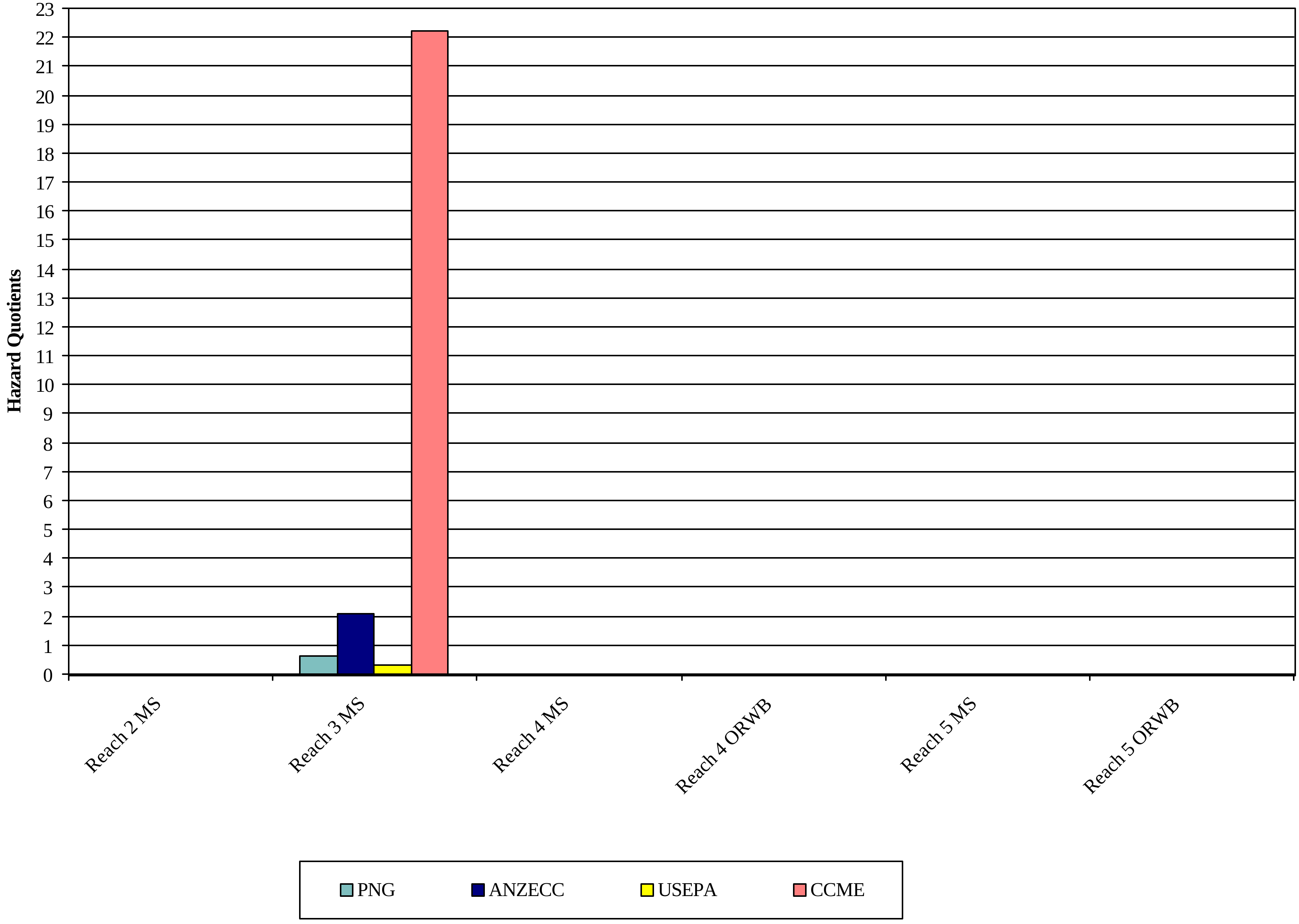


Figure 30. Copper Hazard Quotients for Aquatic Life: Chronic Surface Water.

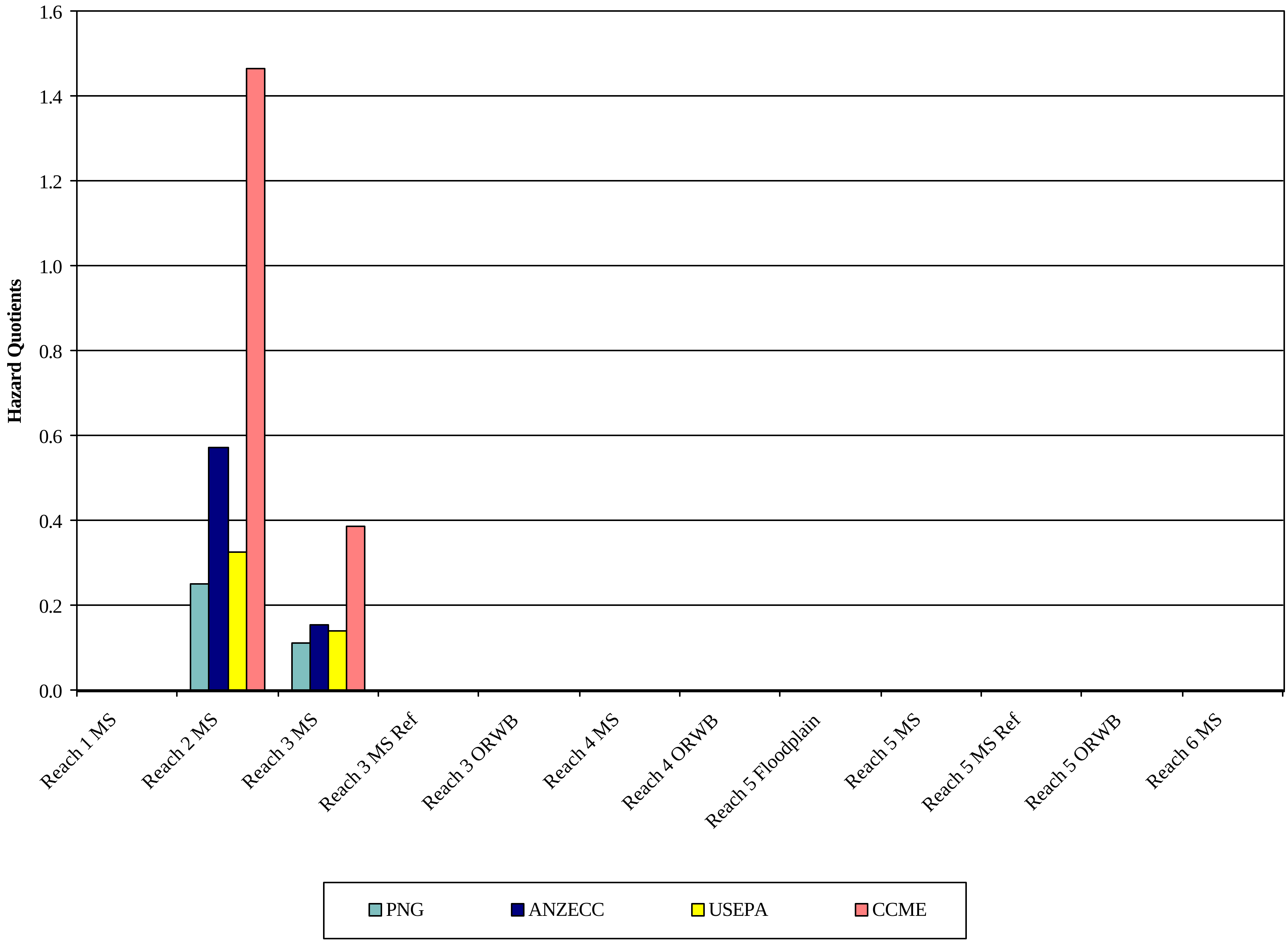


Figure 31. Iron Hazard Quotients for Aquatic Life: Chronic Surface Water.

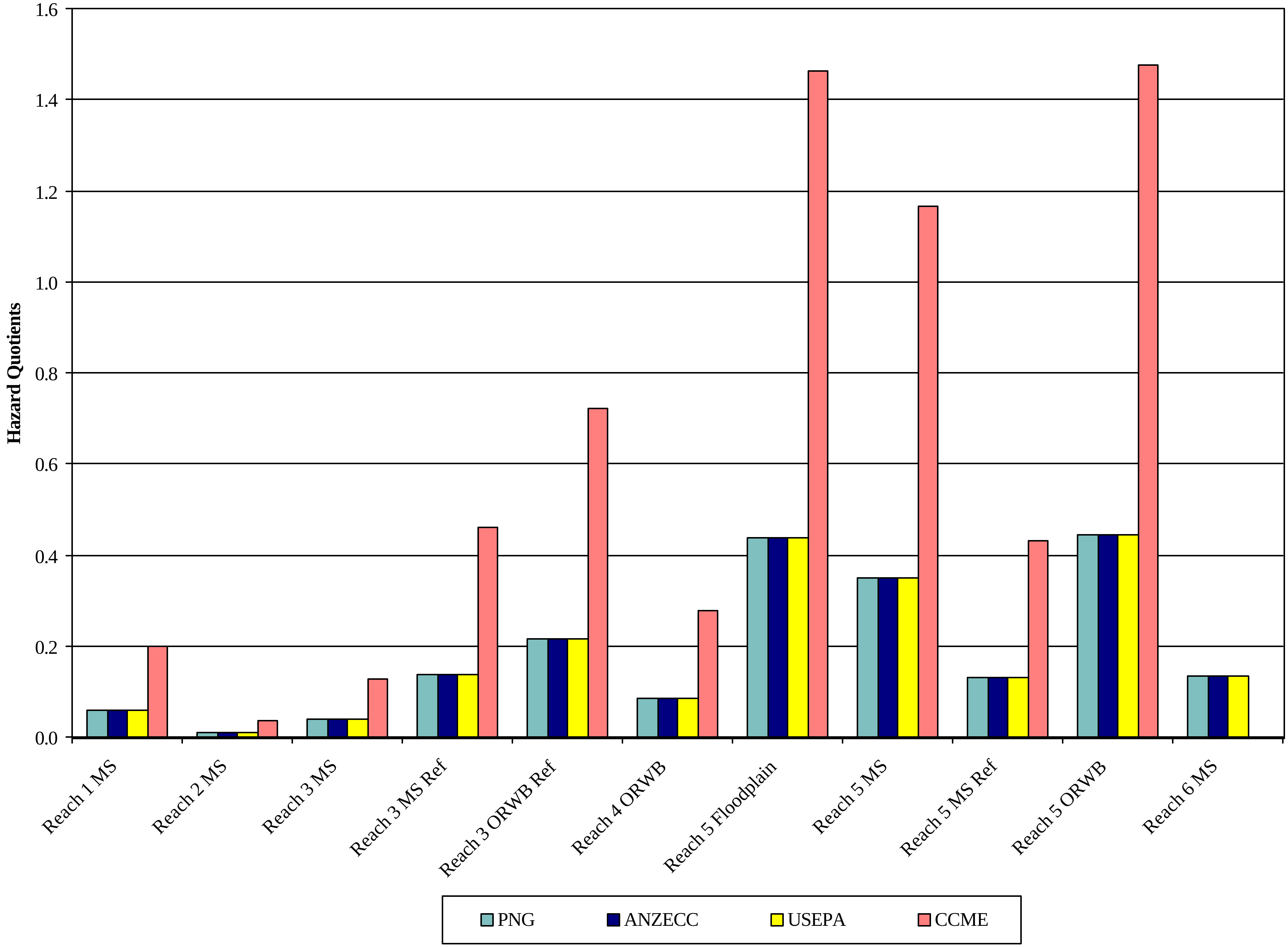


Figure 32. Lead Hazard Quotients for Aquatic Life: Chronic Surface Water.

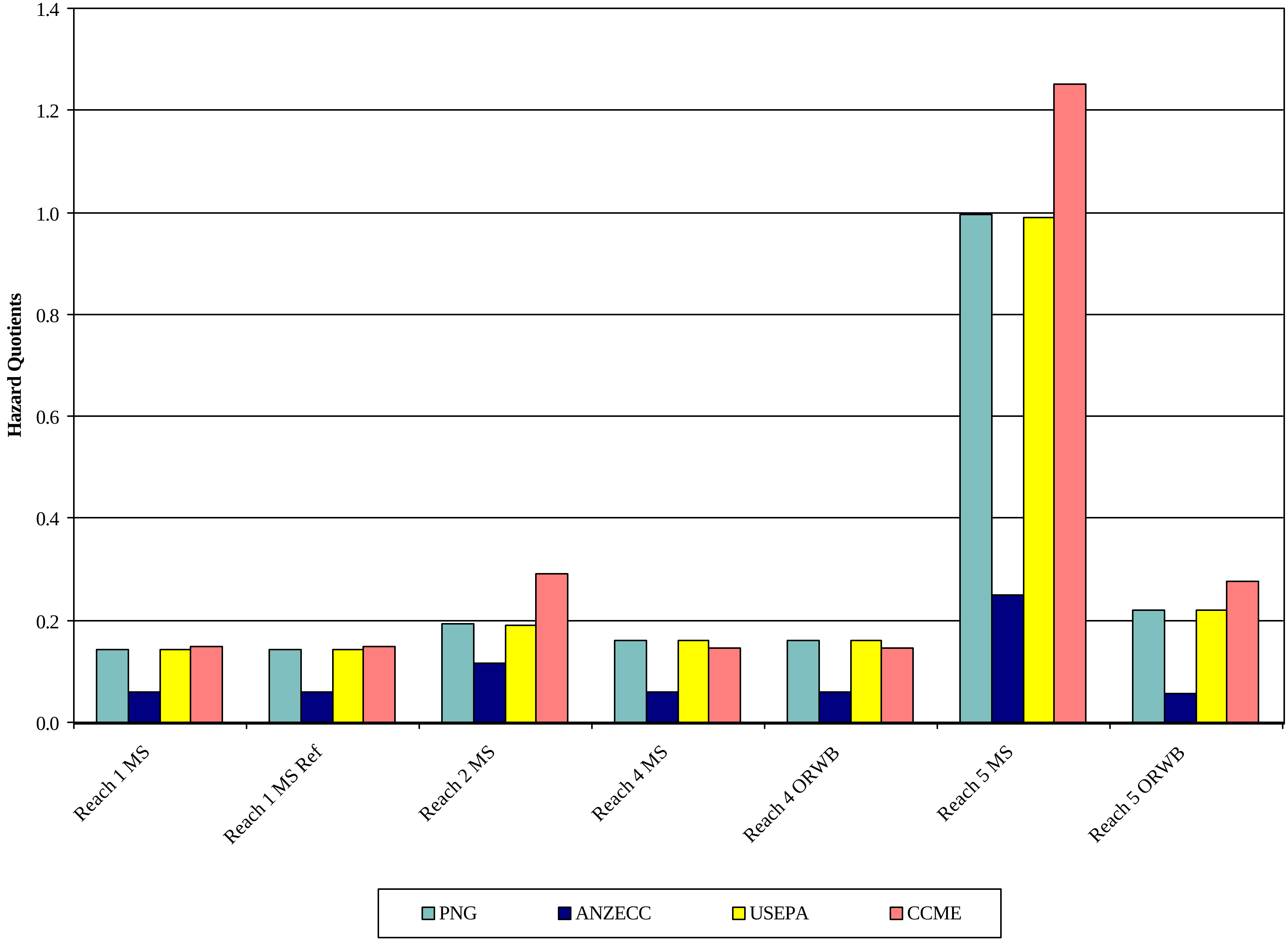


Figure 33. TSS Hazard Quotients for Aquatic Life: Mainstem Reaches 1 - 4.

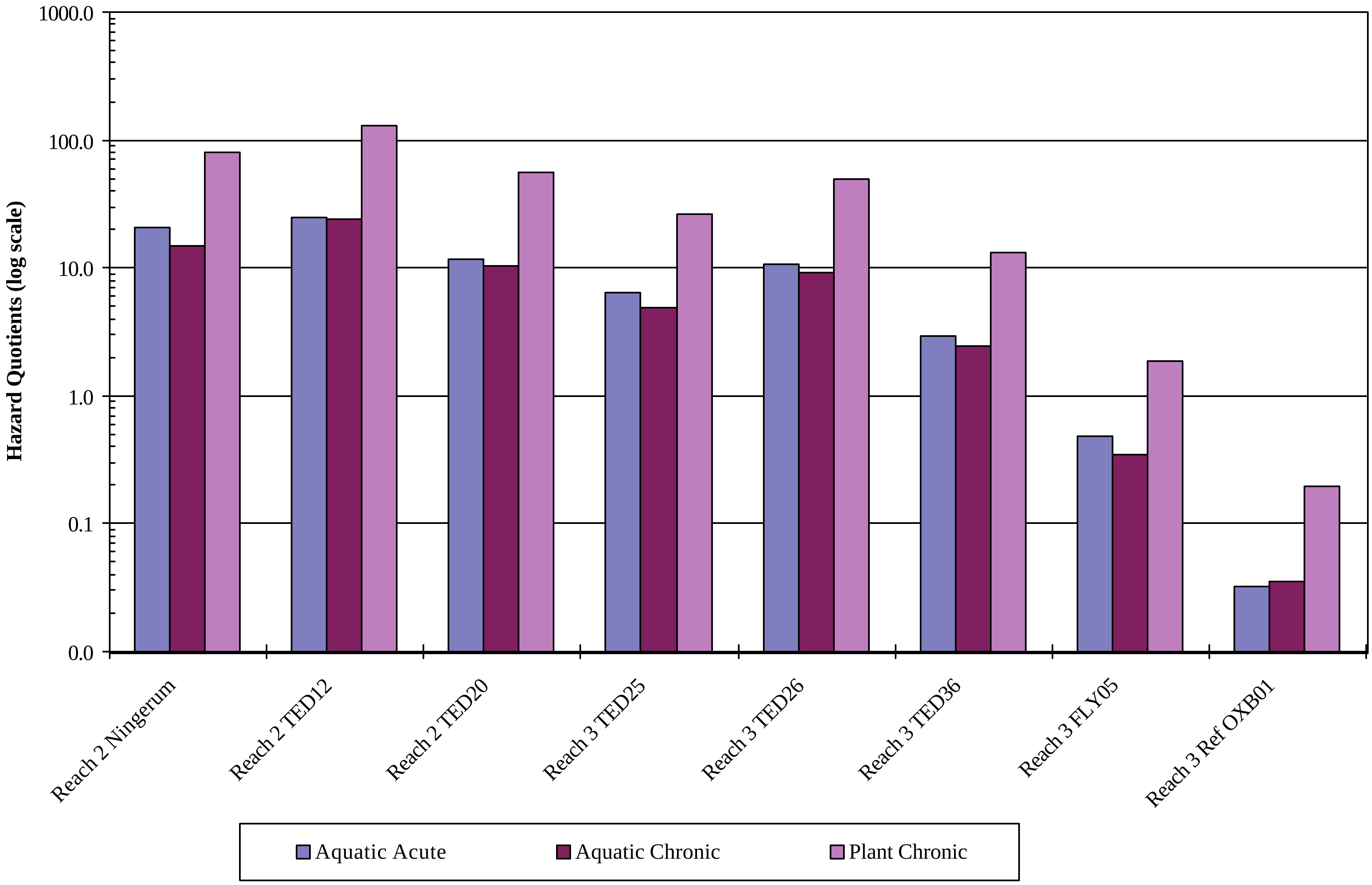


Figure 34. TSS Hazard Quotients for Aquatic Life: Mainstem Reaches 5 - 6.

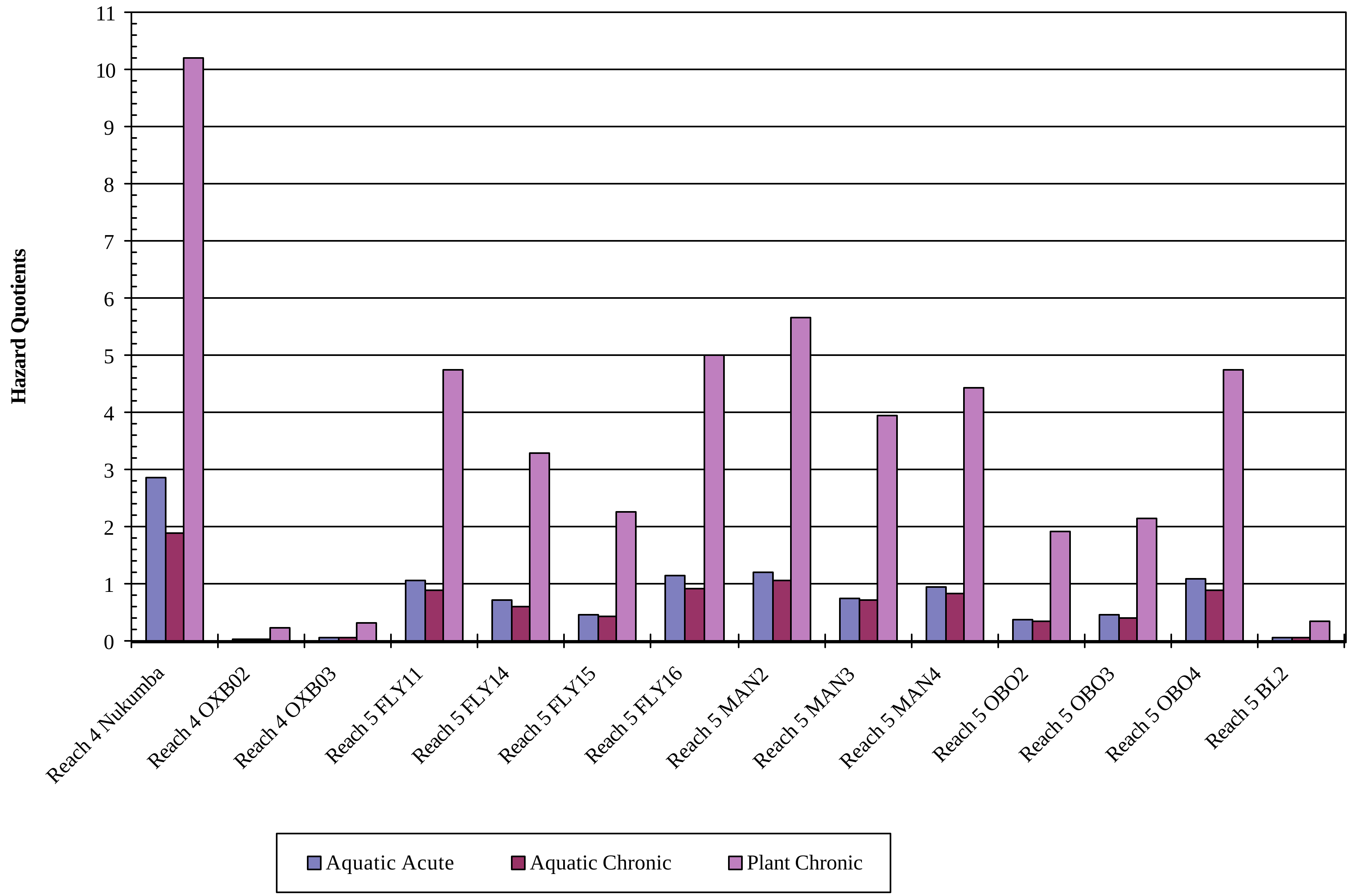


Figure 35. TSS Hazard Quotients for Aquatic Life: Floodplains & ORWBs.

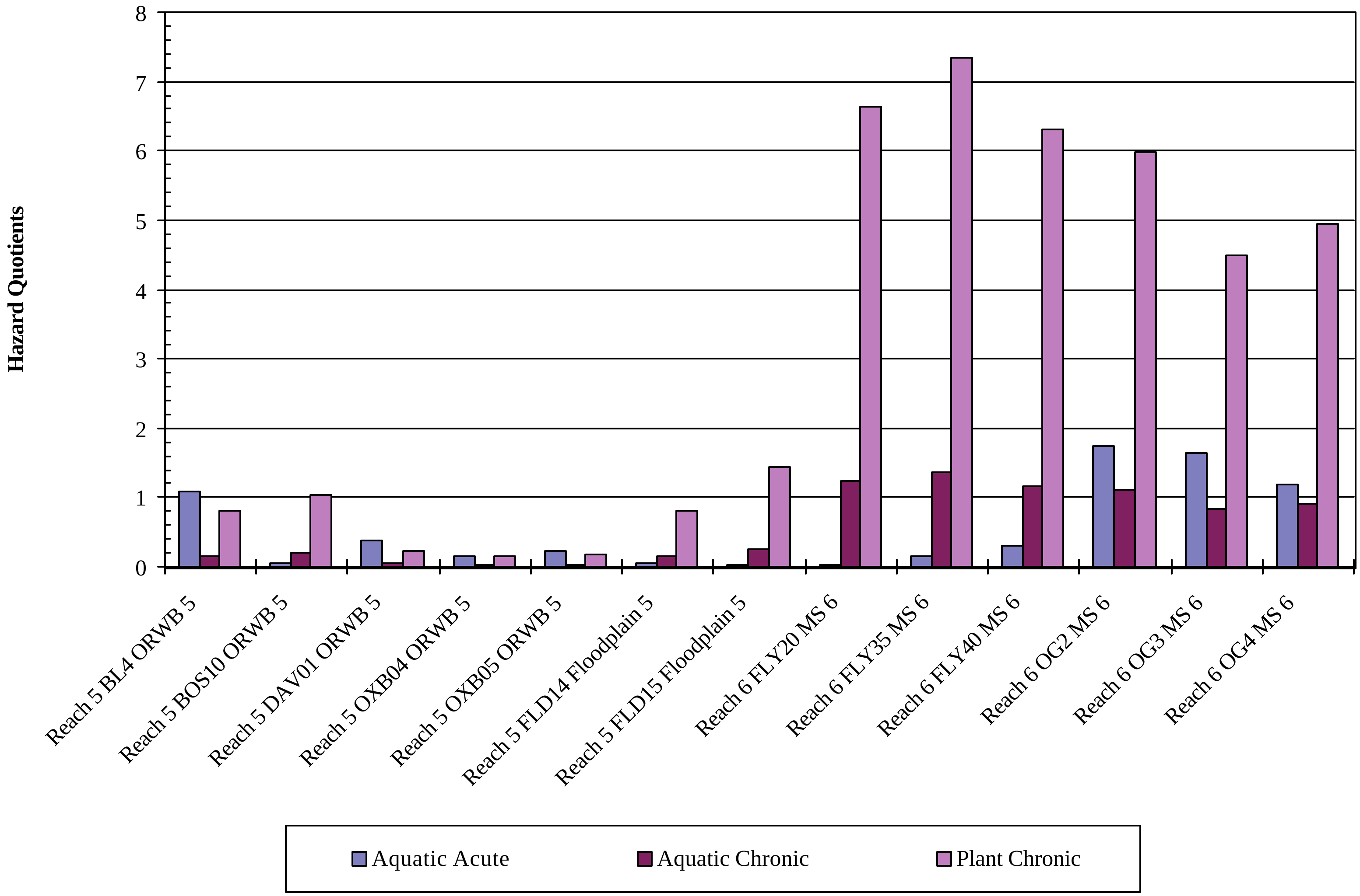


Figure 36. Arsenic Hazard Quotients for Aquatic Life: Chronic Sediment.

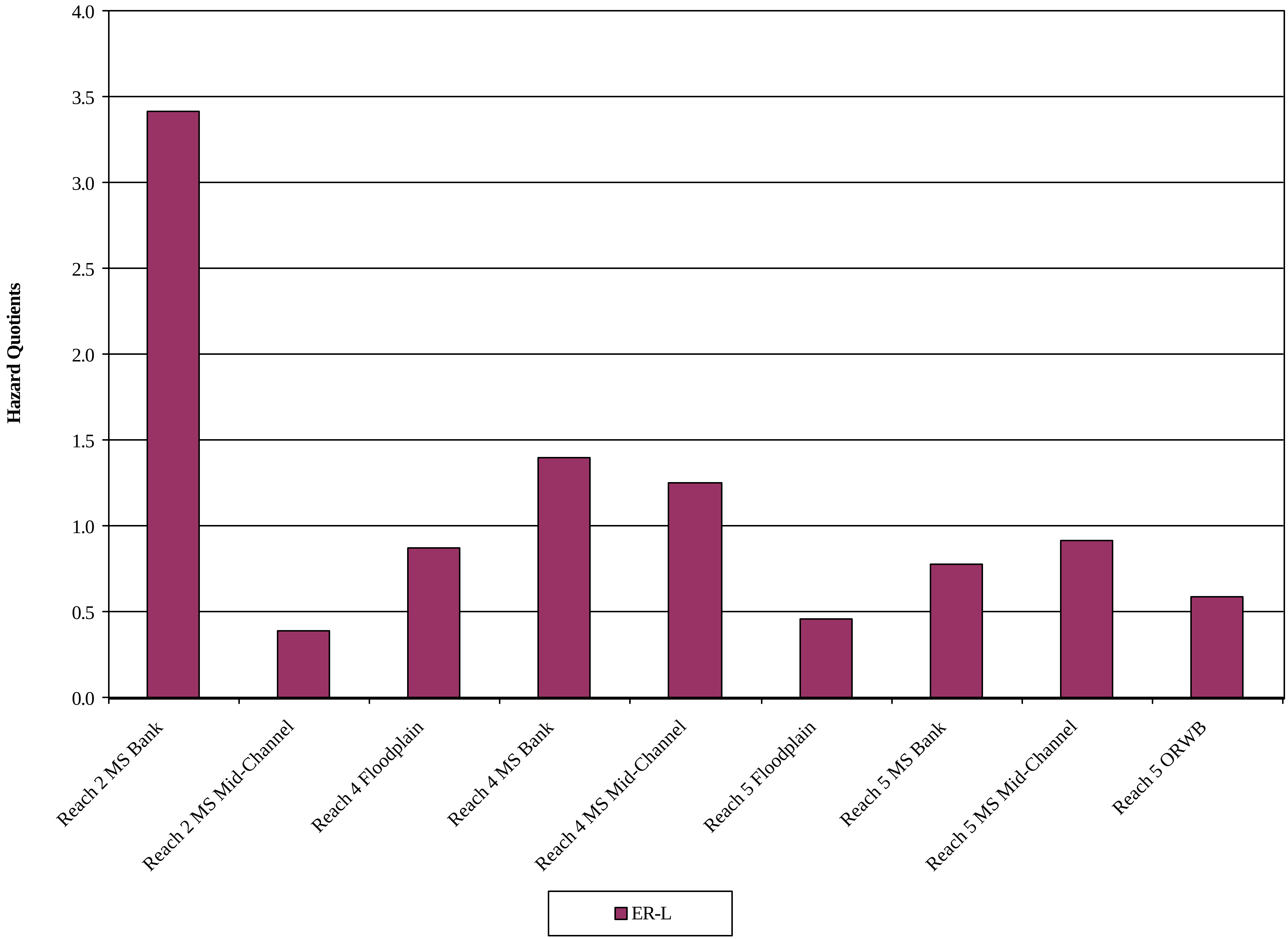


Figure 37. Cadmium Hazard Quotients for Aquatic Life: Chronic Sediment.

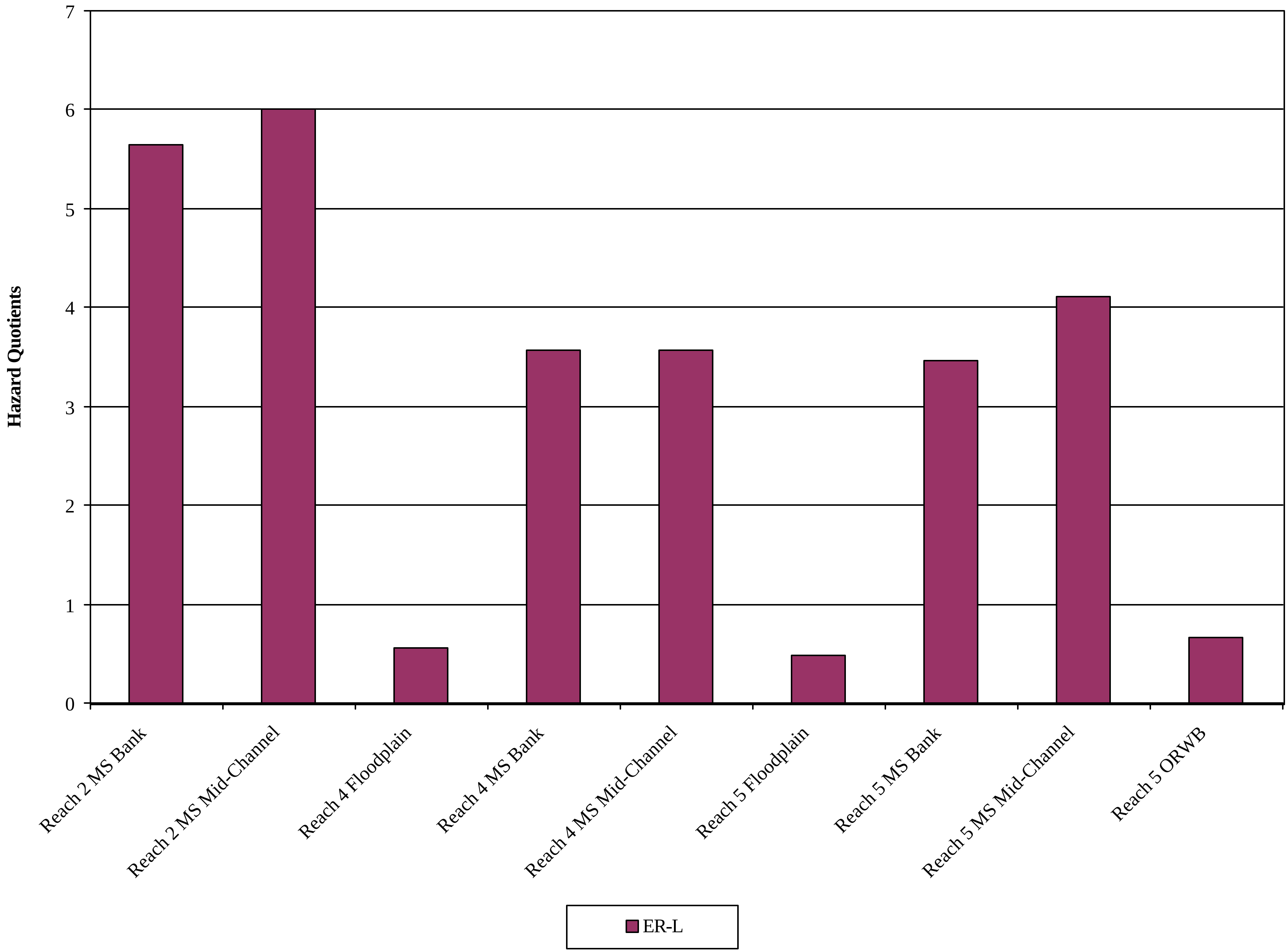


Figure 38. Copper Hazard Quotients for Aquatic Life: Chronic Sediment.

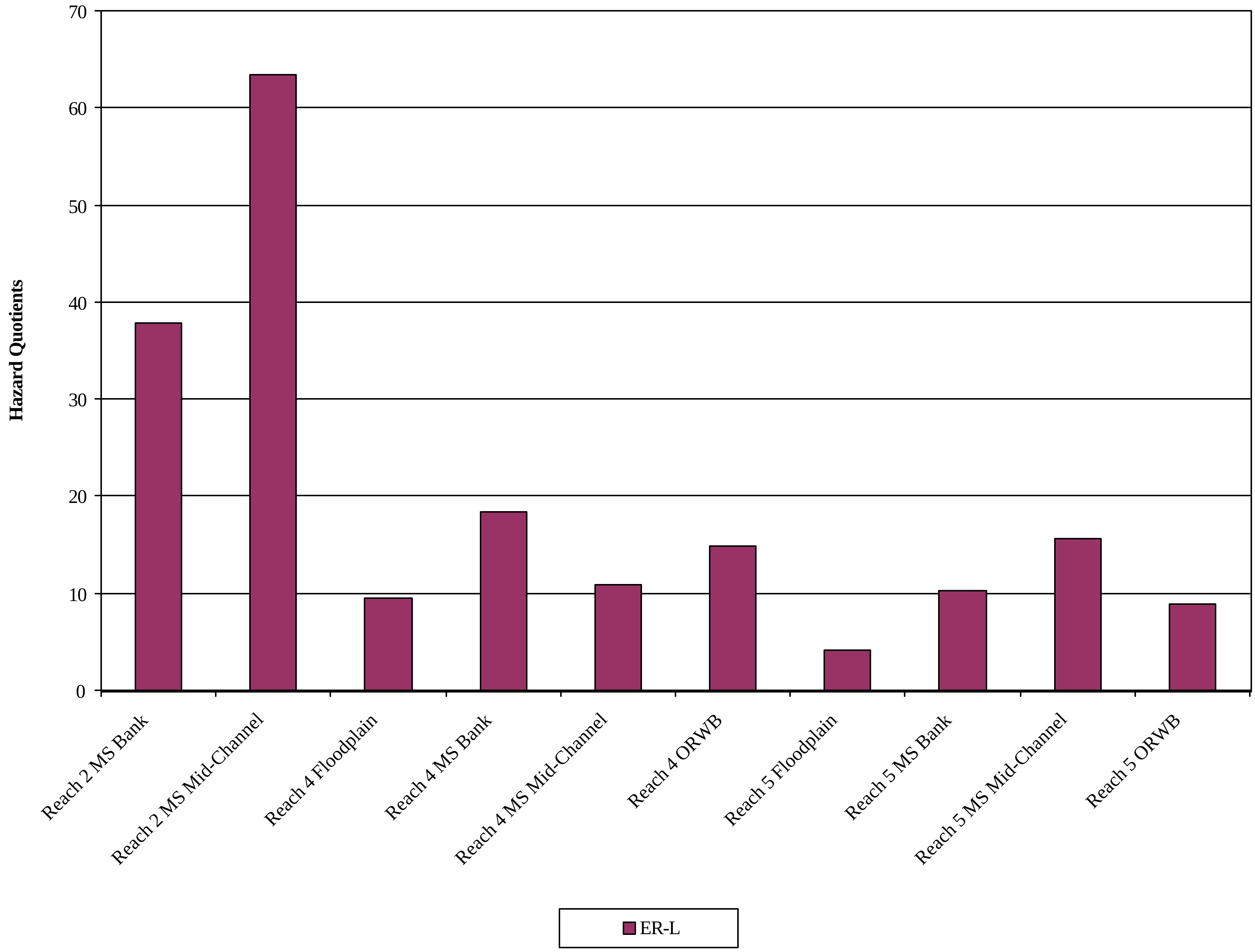


Figure 39. Lead Hazard Quotients for Aquatic Life: Chronic Sediment.

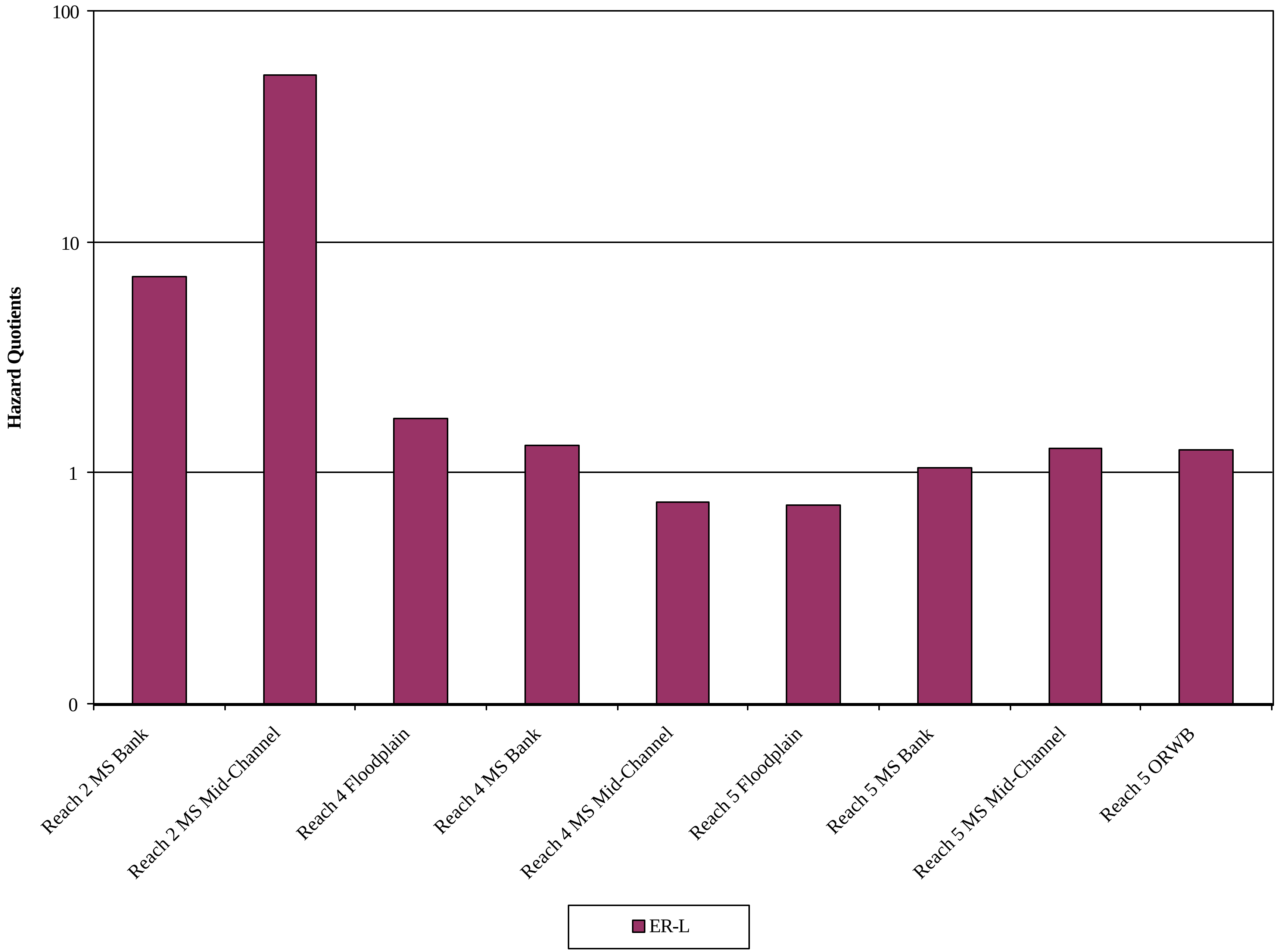


Figure 40. Manganese Hazard Quotients for Aquatic Life: Chronic Sediment.

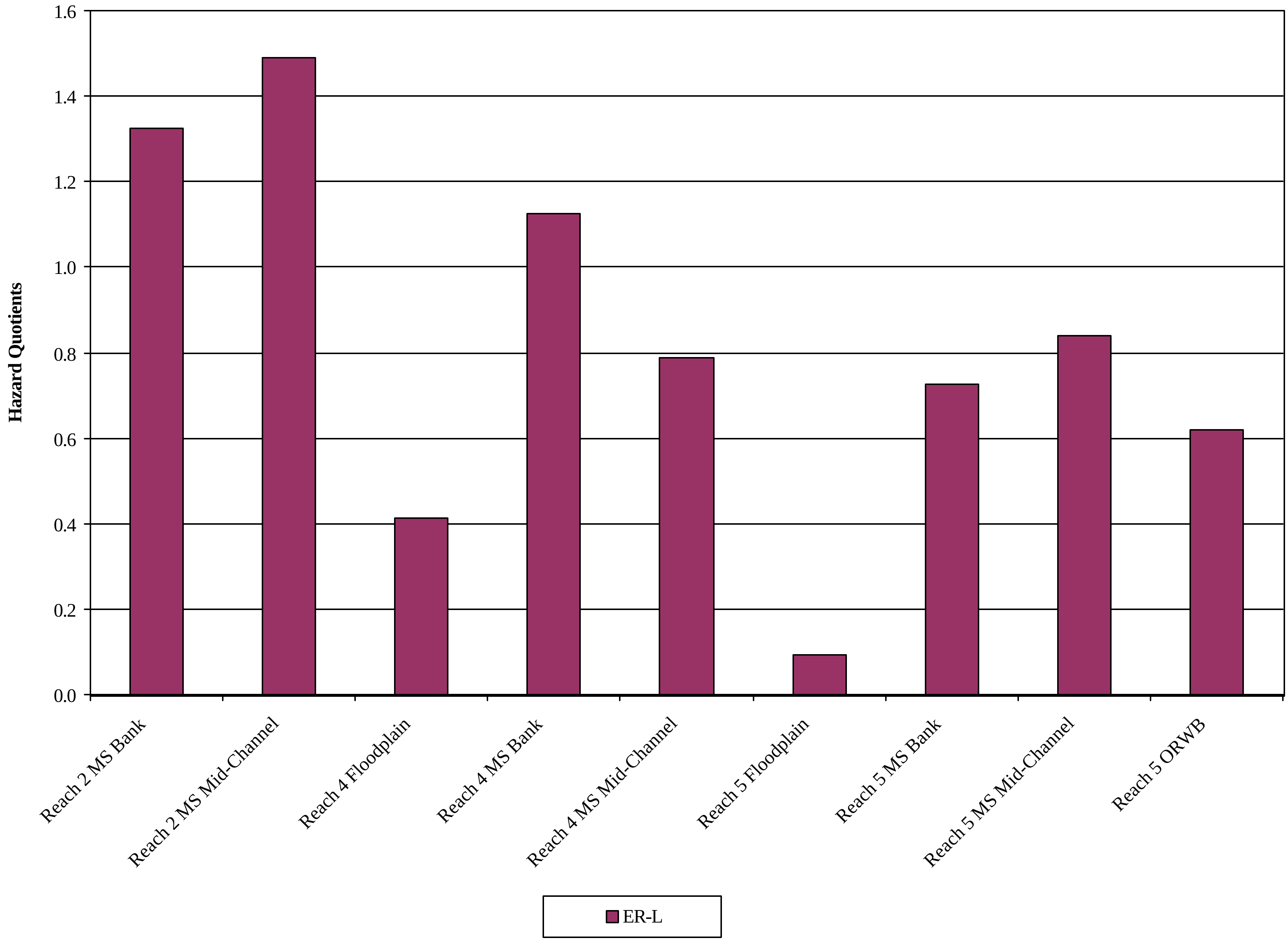


Figure 41. Silver Hazard Quotients for Aquatic Life: Chronic Sediment.

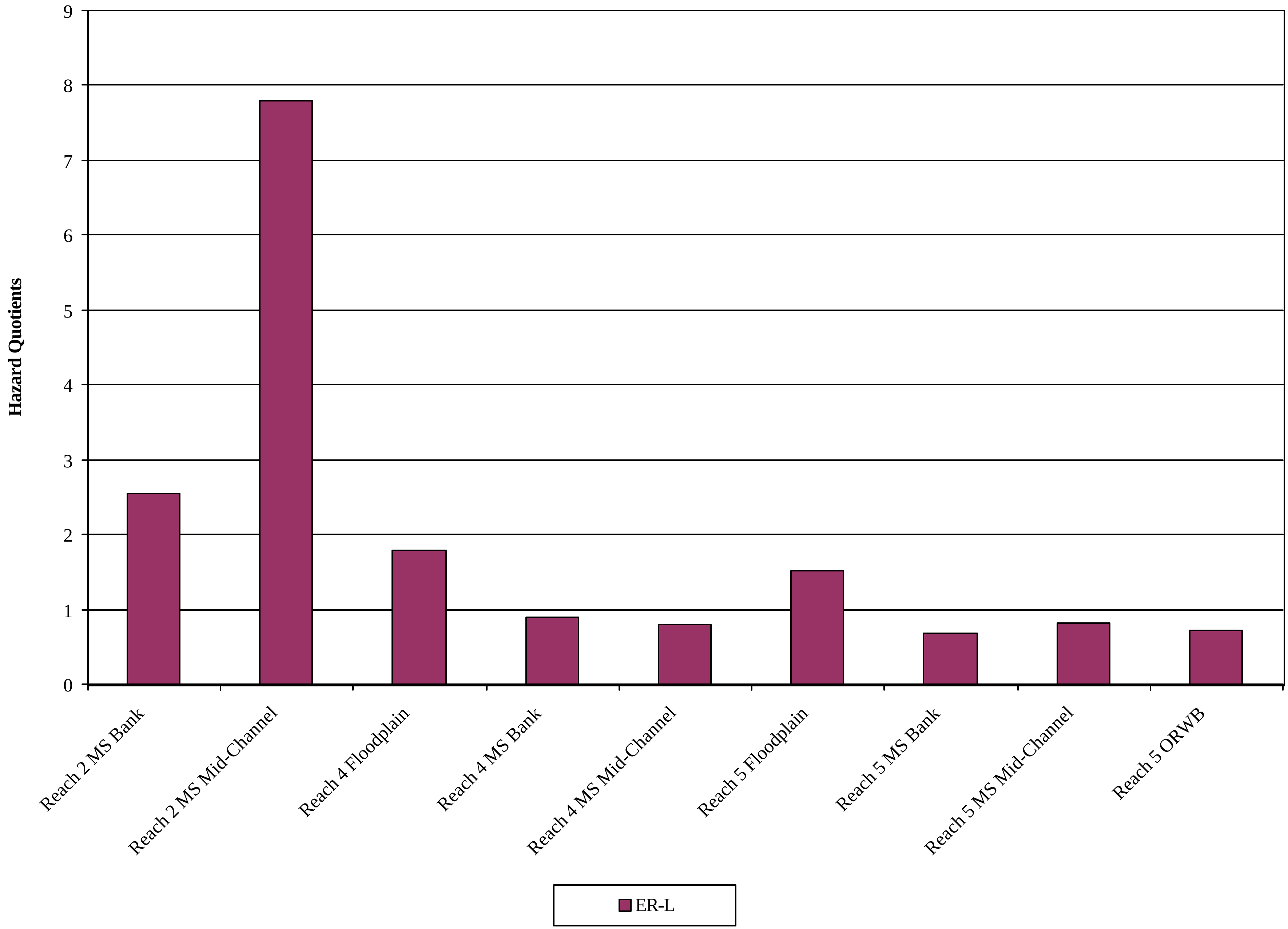
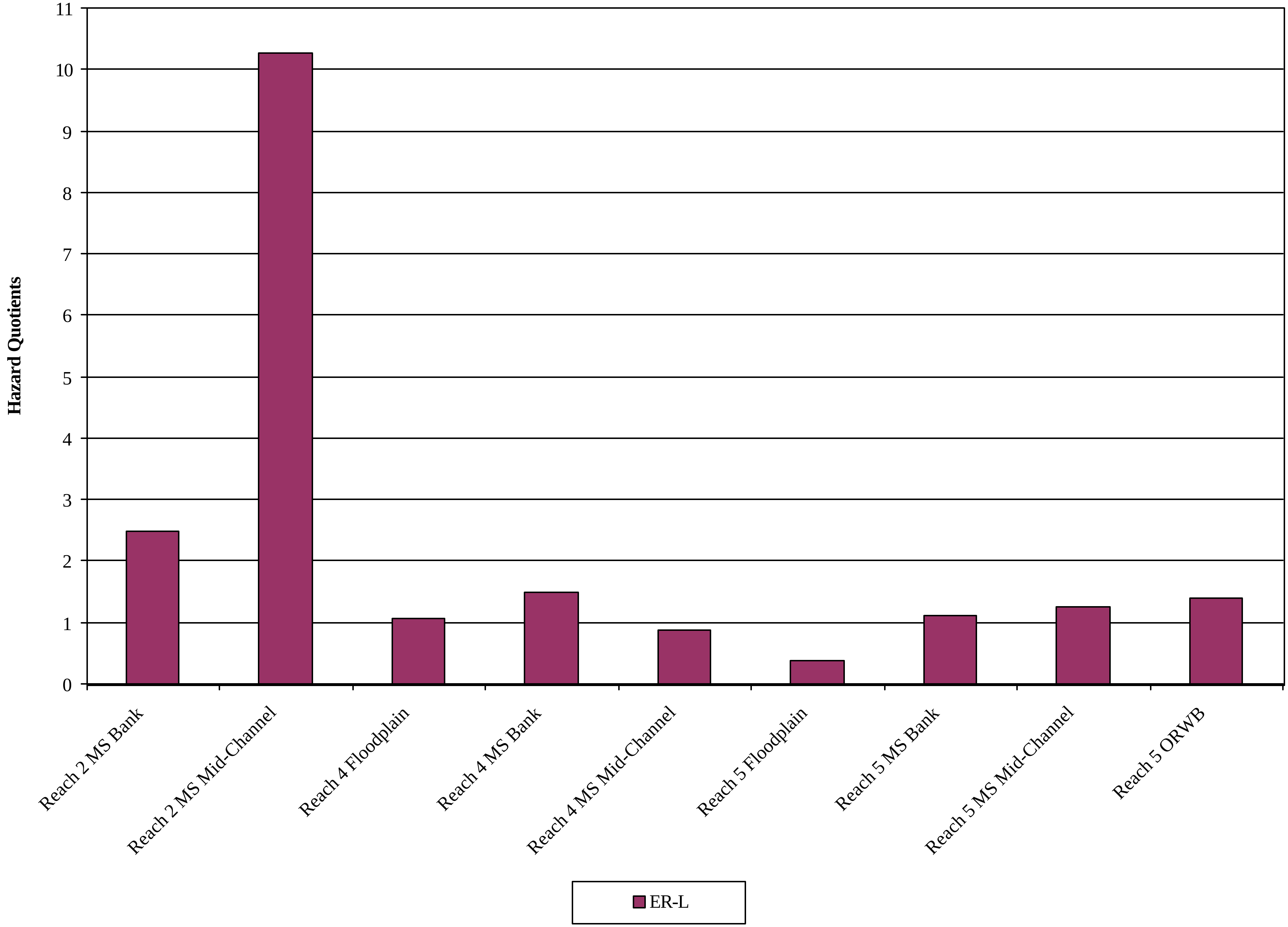


Figure 42. Zinc Hazard Quotients for Aquatic Life: Chronic Sediment.



Freshwater invertebrates in the Fly River have not been monitored extensively enough to provide empirical evidence that they are at risk. Estuarine invertebrates, including crabs and prawns, have been monitored in the Fly River estuary. There have been no detectable mine-induced changes in mud crab or banana prawn populations in the Fly River estuary (OTML 1998a). However, sampling of the reference site, the Bamu River estuary, was discontinued in May 1996, so this comparison is no longer available.

5.4 WILDLIFE

5.4.1 Calculation of Hazard Quotients

For all intake calculations, chemical stressors, and river reaches; a hazard quotient was calculated by dividing the calculated dose or direct exposure for the wildlife receptor by the wildlife toxicity threshold for each chemical stressor. The calculation is shown below and the result is a HQ. Wildlife HQs based on dose were calculated using Equation 13.

$$HQ = \frac{\text{Sum of Forage, Water Dose and Incedental Sediment Dose}}{\text{Soil Invertebrate Toxicity Threshold}} \qquad (13)$$

Soil invertebrate HQs based on direct exposure were calculated using Equation 14.

$$HQ = \frac{\text{Expected Environmental Concentration in Soil}}{\text{Soil Invertebrate Toxicity Threshold}} \qquad (14)$$

5.4.2 Stressors of Potential Concern

Based on the SLRA results, none of the SOPCs evaluated present significant risk to the cassowary, estuarine crocodile, freshwater crocodile, great egret, herbivorous turtle, scavenging turtle, rusa deer or wild pig. Risks, however, were predicted for the fruit bat, white-headed stilt, and terrestrial invertebrates. The following text discusses these results, which are summarised in Tables 30 and 31, as well as Figures 43-45. All HQ calculations for wildlife can be found in Appendix D.

Predicted risks to the fruit bat were from arsenic (HQ = 12), iron (HQ = 29), and lead (HQ = 6.7) exposure via ingestion of food in Reach 2, and copper exposure via ingestion of food in Reaches 2, 4, and 5 (HQs of 17, 3.3, and 1.5, respectively). Risks were also predicted for aluminium in Reaches 2, 4, and 5. However, risks for these metals were higher at background locations and so are not considered of concern. These risk predictions are based on a number of assumptions including the use of main stem or floodplain sediment data as a surrogate for soil data and use of bioaccumulation factors (BAFs) to predict metal concentrations in fruit. If additional data becomes available to reduce uncertainty in the exposure scenario, HQs could be revised or this issue could be further evaluated in a DLRA.

The potential risk for the white-headed stilt due to exposure to copper, lead and zinc was primarily from incidental sediment ingestion. HQs for copper, lead and zinc were 1.0, 3.2, and 2.3, respectively. These risks appear substantive enough to warrant further investigation in a DLRA. Exposures to aquatic invertebrates were based on either a BCF or sediment BAF based approach from the literature and may not be appropriate for site-specific conditions. There also were risks predicted in Reach 5 for the white-headed stilt for aluminium. However, risk from exposure to aluminium at background sites was higher.

Finally, risks from copper, iron, lead, manganese, and zinc were predicted for terrestrial invertebrates via direct contact with soil in Reach 2, and from copper in Reaches 4 and 5. As with risks for other receptors, floodplain sediment data were used as a surrogate for soil data. Although risks were also predicted for aluminium and chromium, HQs for background sites were higher. Therefore, aluminium and chromium were not considered as being of concern for terrestrial invertebrates. The magnitude of these HQs suggest that terrestrial invertebrates are at risk. However, there are uncertainties in these risk estimates that are discussed in Section 5.5.4.

Table 30. Wildlife HQs for aquatic pathways.

Stressor	Reach	Location	HQ	Receptor	Primary Source of Exposure
Copper	4	Off river water bodies	1.0	Stilt	Incidental sediment ingestion
Lead	5	Off river water bodies	3.2	Stilt	Incidental sediment ingestion
Zinc	5	Off river water bodies	2.3	Stilt	Incidental sediment ingestion

5.4.3 Field Evidence of Risk to Wildlife

Effects on wildlife may occur from exposure to chemical stressors, and loss of habitat and food resources. However, there are no wildlife survey data available to evaluate the evidence of adverse effects on wildlife. There is evidence, however, of effects on terrestrial vegetation (Duff 1992) that provide habitat for many wildlife species (Section 4.2.1.1). Furthermore, there is evidence of the loss of fish (reduction in biomass) in the main stem of the Fly River from Reach 2 to Reach 5 (Section 5.3.2). These two non-chemical effects, loss of habitat and loss of food resources, may be affecting wildlife along with chemical stressors.

Figure 43. Copper, Arsenic, Iron and Lead Hazard Quotients for Fruit Bats.

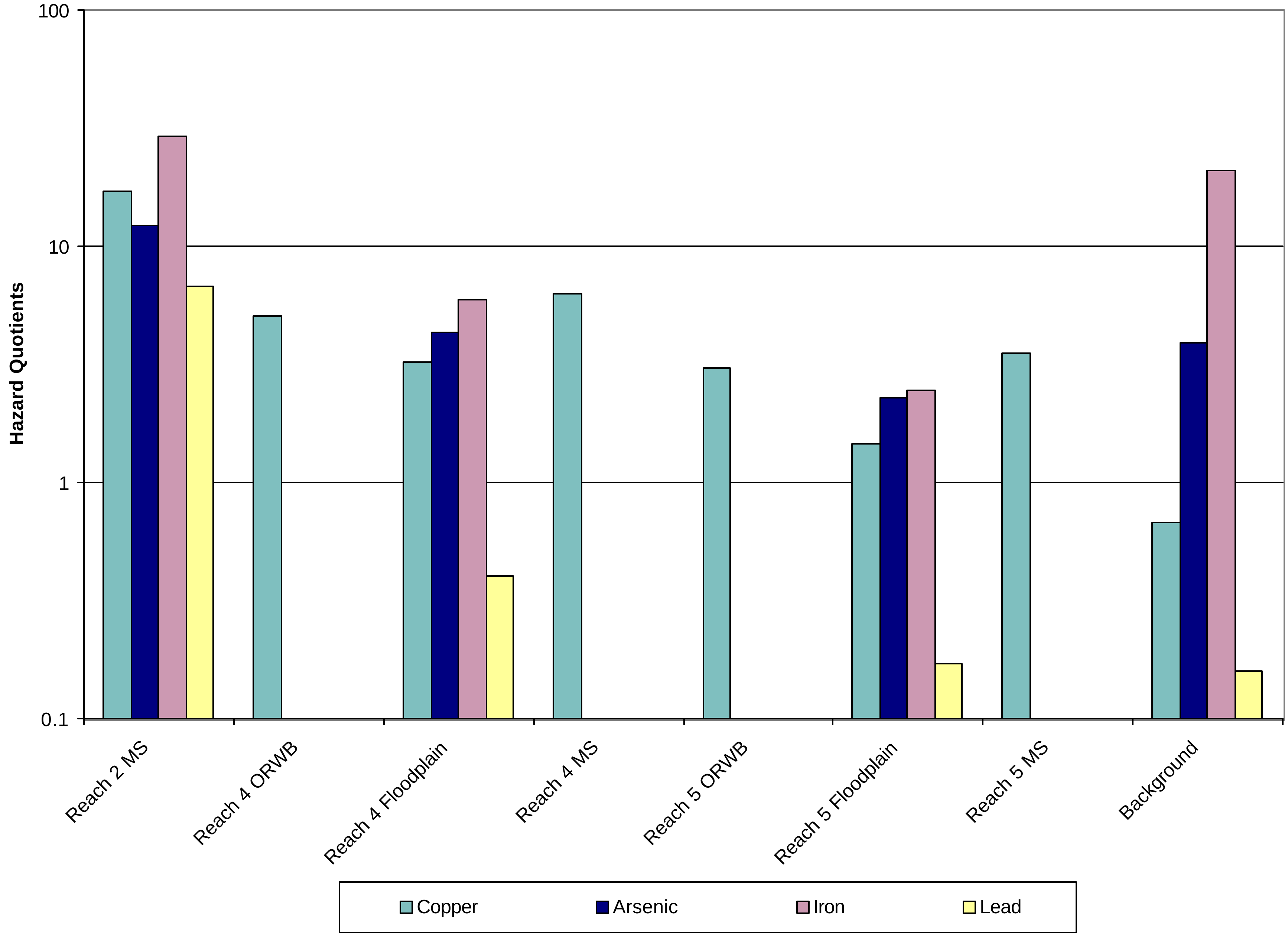


Figure 44. Copper, Iron, Lead, Manganese and Zinc Hazard Quotients for Terrestrial Invertebrates.

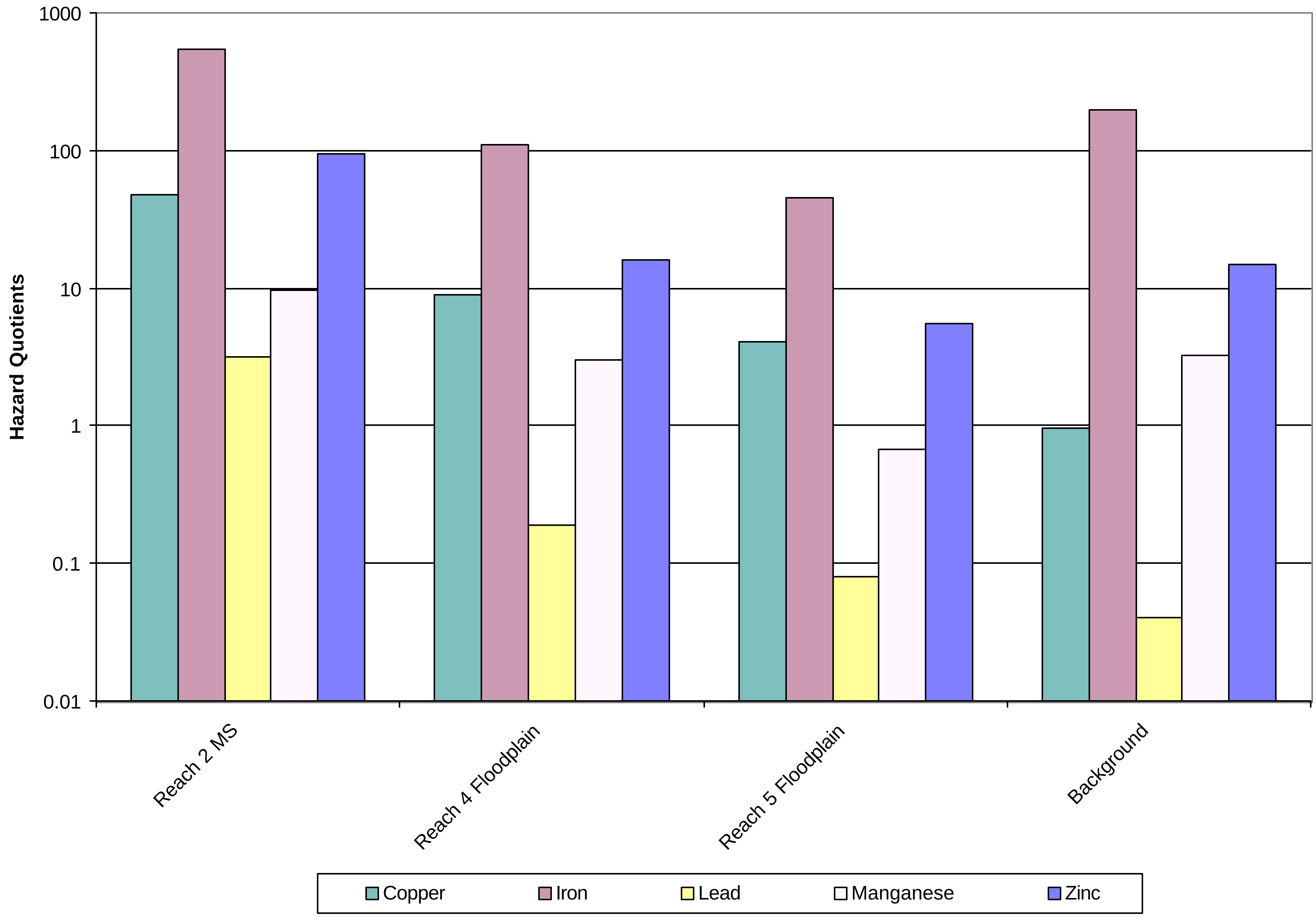


Figure 45. Copper and Lead Hazard Quotients for White-headeded Stilts.

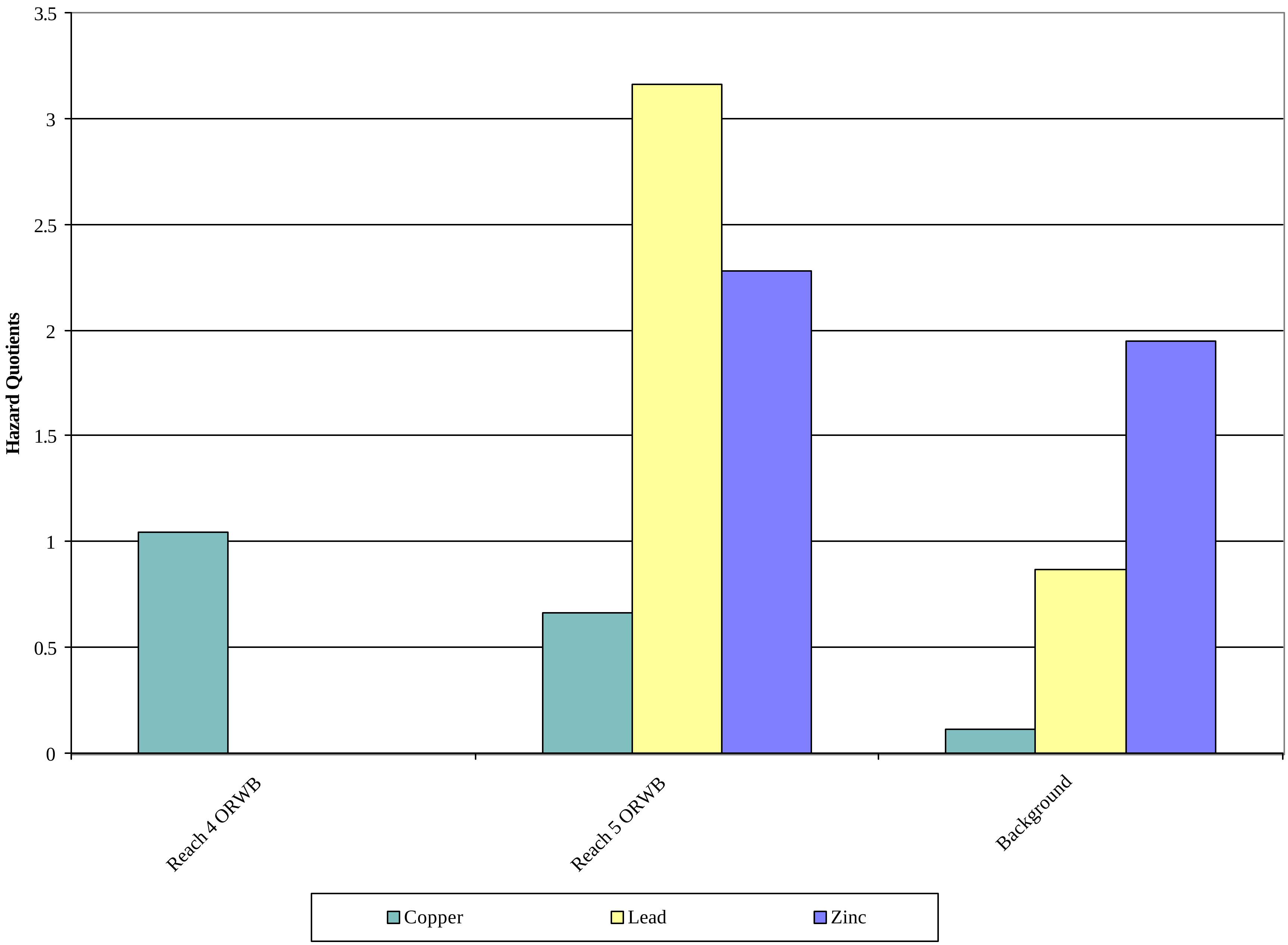


Table 31. Wildlife HQs for terrestrial pathways.

Stressor	Reach	Location	HQ	Receptor	Primary Source of Exposure
Copper	2	Main stem	47	Terrestrial invertebrate	Direct soil contact
	4	Flood plain	9.1	Terrestrial invertebrate	Direct soil contact
	5	Flood plain	4.0	Terrestrial invertebrate	Direct soil contact
Iron	2	Main stem	544	Terrestrial invertebrate	Direct soil contact
Lead	2	Main stem	3.2	Terrestrial invertebrate	Direct soil contact
Manganese	2	Main stem	9.8	Terrestrial invertebrate	Direct soil contact
Zinc	2	Main stem	94	Terrestrial invertebrate	Direct soil contact
Arsenic	2	Main stem	12	Fruit bat	Food ingestion
Copper	2	Main stem	17	Fruit bat	Food ingestion
	4	Flood plain	3.3	Fruit bat	Food ingestion
	5	Flood plain	1.5	Fruit bat	Food ingestion
Iron	2	Main stem	29	Fruit bat	Food ingestion
Lead	2	Main stem	6.7	Fruit bat	Food ingestion

5.5 UNCERTAINTIES

An important component of any risk assessment is a discussion of the uncertainties associated with the assessment. Typically, the major uncertainties in an HERA are associated with the representativeness of the exposure data (in terms of concentration and accurately reflecting chemical bioavailability) and the applicability of the effects thresholds to site-specific conditions. For the SLRA, the greatest uncertainty was associated with lack of exposure data for a number of SOPCs. As a result, the SLRA does not adequately assess risks resulting from mine waste discharge, but rather only certain components within the waste.

Specific uncertainties for each component of the SLRA are described in the remainder of this section.

5.5.1 Human Health

The major uncertainties in the human health risk assessment are identified below.

1) Doses exceeding threshold criteria were predicted for cadmium and lead. At the time of this report, no biomonitoring data (e.g., blood lead) were available to validate the likely significance of these exceedances. Hair data are available for the Ok Tedi and Fly Rivers. However, hair data are not well correlated with potential adverse effects and no hair data from background locations are available to assess the significance of concentrations in hair that exceed literature-based values. For arsenic, no background data were available, preventing an assessment of background cancer risk potential to put estimated exposures in perspective;

2) Data in tissue was limited to four chemical stressors, cadmium, lead, copper, and zinc. Other stressors which are significant for human health (e.g., methylmercury) were not available;

3) The human health exposure assessment is based on conservative (i.e., health protective) assumptions regarding the frequency, duration, and extent of contact with affected environmental media. Thus, the risk predictions are likely biased towards over-prediction. Site-specific information concerning these parameters would allow further refinement of the dose estimates;

4) The rate of consumption of aquatic resources (i.e., fish and shellfish) and other foods (e.g., sago and cassava) from affected rivers, the Fly River estuary or other mine affected-harvest locations is not well characterised and the representativeness of fish species sampled to those eaten by local people in the Ok Tedi or Fly Rivers and estuary is not known; and

5) Children are potentially more sensitive to chemical stressors than adults. The exposure characterisation and effects characterisation were based on a lifetime exposure integrating risks to both children and adults. A secondary assessment was performed to evaluate potential risks to children independently. Details of this assessment are described below.

5.5.1.1 Potential Child Specific Risks

The sensitivity of people to chemicals depends on many factors, including their age. Chronic (long-term) non-cancer health risks were quantified based on a life-time, weighted chemical intake, which incorporated the differing ingestion rates and bodyweights that occur throughout various life stages (i.e., child, adult). However, to address subchronic[14] risks to children (i.e., < 5 years old) who may be exposed to tailings from a period of months to several years, separate hazard quotients were also calculated.

This child-specific evaluation needs to be qualified, however. Calculating separate HQs for children generally is inappropriate (overly conservative even for a SLRA) because it involves comparing short-term (subchronic) exposures to toxicity values that were derived considering long-term (in many cases, lifetime) exposures. Because of this, child-specific HQ calculations do not hold the same weight as the calculations based on lifetime weighted average exposures. Nevertheless, such comparisons have been made to determine whether HQs for children might exceed the screening threshold of 1. If an HQ exceeded this threshold, further evaluation was undertaken to determine whether this chemical could present a possible risk to younger age groups based on the scientific evidence.

Sago and cassava data have been identified for Reaches 3, 4, and 5. Therefore, HQs exceeding 1 for sago consumption are discussed for these reaches. Chemical stressors that exceeded their

[14] Subchronic refers to exposures lasting less than a lifetime or many years. Here it refers to exposures lasting months to a few years.

toxicity values for all other exposure pathways are presented in Table 32. The likely significance of each of these exceedances is discussed further below.

Table 32. Potential child-specific SOPCs.

Location	Site Type	Chemical	Exposure Pathway (s) [a]	Site HQ	Reference HQ
Reach 2	Main Stem	Iron	Sediment, Surface water contact [b]	4.0	ND
Reach 3	Main Stem	Chromium	Sago Consumption	1.6	ND
Reach 3	Main Stem	Manganese	Sago Consumption	2.0	ND
Reach 3	Main Stem	Nickel	Sago Consumption	2.5	ND
Reach 3	Main Stem	Cadmium	Aquatic Food Consumption	3.0	< 0.01
Reach 4	Off River Water Bodies	Cadmium	Aquatic Food Consumption [c]	0.93	ND
Reach 4	Main Stem	Chromium	Sago Consumption	1.6	ND
Reach 4	Main Stem	Manganese	Sago Consumption	2.0	ND
Reach 4	Main Stem	Nickel	Sago Consumption	2.5	ND
Reach 4	Main Stem	Iron	Sediment Contact [b]	1.3	ND
Reach 5	Off River Water Bodies	Aluminium	Sediment Contact	1.6	ND
Reach 5	Main Stem	Chromium	Sago Consumption	1.6	ND
Reach 5	Main Stem	Manganese	Sago Consumption	2.0	ND
Reach 5	Main Stem	Nickel	Sago Consumption	2.5	ND
Reach 5	Main Stem	Lead	Aquatic Food Consumption [c]	0.82	0.75
Estuary		Cadmium	Aquatic Food Consumption	9.6	6.8

[a] The pathways shown contribute the majority of the dose, and therefore most of the potential risk.
[b] HQs exceeding 1, are based on cumulative exposure to sediment and/or surface water from all pathways.
[c] The pathway shown does not, by itself, result in an HQ greater than 1. However, it does contribute the majority of the dose and cumulative HQ.
ND -no data for this medium and location.

With the exception of cadmium, the child-specific HQs greater than one for all chemicals in Table 32 range from slightly greater than 1 to 4. Aluminum slightly exceeded its TRV in Reach 5. However, aluminum is reported to have low toxicity, and the toxicity threshold is based on a level that will prevent aluminum precipitates and discoloration in treated water (WHO 1996a). Currently available evidence does not support a causal role for vhronic exposure to aluminum in any adverse health effets, ecept in cases of kidney failure (IPCS 1997; Klien 1990).

Iron HQs for children exceeded one in Reach 2 and Reach 4 based on primarily sediment exposures (Reaches 2 and 4) and surface water (Reach 2). However, these risks are likely to be significantly lower than predicted for several reasons. First, iron is an essential nutrient and generally has very low toxicity (see Section 4.1.1). Second, the surface water HQ in Reach 2 similarly assumed complete absorption following incidental iron ingestion; however, in particulate forms iron will be considerably less bioavailable. Finally, for iron, estimates of the minimum daily requirement range from 10-50 mg/day (WHO 1996a); however the estimated incidentally ingested intake (i.e., total iron) equates to only 14 mg/day.

Consequently, the actual risk associated with incidental ingestion of iron in Ok Tedi river water is likely considerably less than calculated. Finally, the toxicity value for iron is based on excessive iron accumulation in tissue (WHO 1996a), though it has not been established in the scientific literature that iron accumulation in human tissue is associated with any adverse health effects.

Cadmium hazard quotients exceeded one in Reaches 3, 4, and the estuary. In Reaches 3 and 4 the HQs are attributable to aquatic food consumption and these results parallel those predicted for lifetime exposure. In Reach 4, the HQ results from cumulative exposure by several pathways, the greatest dose coming from aquatic foods. However, calculation of cadmium risks for children is overly conservative because, as described in Section 5.1.3, the toxicity value is based on an adverse effect (i.e., excretion of protein in urine) that requires several decades of exposure to acquire. Thus, the application of the cadmium toxicity value, based on decades of exposure, to a short time period is an overly conservative estimation of risks. Thus, cadmium exposure at the levels estimated for a relatively shorter duration of time (i.e., months to a few years) is not expected to have potential for child-specific risks.

A lead hazard quotient exceeding one was predicted in Reach 5 from the cumulative exposure to sediment, aquatic foods, and sago, with the majority of the dose coming from aquatic foods. The magnitude of the exceedance is slight. The risk potential is dependent, in part, on the extent to which sago is consumed because the sago dose is the second largest contribution to the total dose. Although sago consumption is not the dominant source of the dose, the sago concentration data, and the rate at which it may be consumed are relatively more uncertain than data from the other exposure pathways.

Sago data were collected from the lower Ok Tedi and middle Fly Rivers. Therefore, these data were assumed to be representative of Reaches 3, 4, and 5. Chromium HQs exceeded one for children consuming sago. However, risks for chromium may have been over predicted because the toxicity value for chromium was based on the most toxic form, hexavalent chromium (Table 13). In foods, the majority of chromium is likely to be present in the trivalent state (Cr III), as it is the most stable form and forms organic complexes (Kimbrough et al. 1999; Gad 1989). The toxicity value for trivalent chromium is approximately 1000 –fold greater than that for hexavalent chromium. The HQ based on the toxicity value appropriate to the chemical form likely to be present in sago (trivalent) would be 0.0033. Because the chemical form of chromium in sago is likely to be predominantly trivalent there is little concern for the hexavalent chromium exceedances.

Manganese in sago was also identified as posing potential risks to children. Manganese is an essential dietary nutrient (Section 4.1.1) and is considered to be among the least toxic elements when administered orally (WHO 1996b). In general, diets high in unrefined cereals, whole grains, leafy vegetables and tea are high in manganese (WHO 1996b; NRC 1989). Intakes recommended for children (4-6 years old) have been set at 1.5 to 2 mg/day (NRC 1989). Intakes of 20 mg/day can be tolerated without ill effects (WHO 1996a). For comparison, child intakes from sago have been estimated in this risk assessment at 2.8 mg/day. Thus, the estimated intakes are likely to be adequate for proper nutrition but unlikely to have potential for adverse health effects.

Nickel also exceeded its toxicity value. Similar to manganese, nickel occurs commonly in plant foods (WHO 1996b). Little data are available on the amount of nickel in diets but intakes in the U.K. have been reported to range from 0.014 to 0.250 mg/day for children, although diets that are high in unrefined and unprocessed foods are likely higher than this (WHO 1996b). The intakes from sago are estimated in this risk assessment at 12 mg/day. For this screening risk assessment it was assumed that 10% of the nickel consumed would be absorbed. However, inorganic nickel is poorly absorbed across the gastrointestinal tract and absorption has been estimated at less than 10% (Friberg 1986). Finally, the nickel toxicity value has a built in safety factor of 1000 to account for uncertainties in its toxicity (Section 4.1.1). Thus, the nickel toxicity value is very conservative and actual risks are likely to be lower than those predicted due to the very high margin of safety built in to the toxicity value, and the very low magnitude of the exceedance.

In general, the risks predicted for children are not of concern for several reasons. First, the application of chronic toxicity values developed for lifetime exposures to subchronic exposures (i.e., <5 years) is highly conservative, even for a SLRA. Nickel has a very large uncertainty factor of 1000, to provide an added margin of safety. Thus, intakes associated with effects could actually be higher than identified by the conservative toxicity value. As discussed previously, the chromium toxicity value used represents the most toxic chromium species---chromium (VI). However, most chromium in food is generally in the trivalent (III) form. The toxicity threshold for chromium (III) is ~1000 times higher than the value used, so the chromium exceedance is not likely to be associated with health effects. The cadmium exceedances are not expected to pose health concerns for children because the health effect of concern is based on decades of exposure at levels exceeding the toxicity value. Finally, two of the chemicals identified (iron and manganese) are essential dietary nutrients found in food at varying levels and their predicted intakes for children are within the nutritional range. The toxicity values based on adult intake and body weights also do not take into account the higher dietary requirements per body weight for children. Thus, dietary requirements for children may in some cases approach or exceed toxicity values. These factors suggest that the exceedances predicted would not result in health effects.

5.5.2 Terrestrial Vegetation

5.5.2.1 Physical Stressors

There are many sources of uncertainty associated with the screening phase of the HERA (i.e., the SLRA) for physical stressors on terrestrial vegetation. While the general conceptual models of exposure appear valid, there is only fragmentary data on sublethal stressor effects for many of the receptors. This is largely because of the extreme diversity of the potentially affected vegetation types. While flooding is known to produce dieback, it is not known how long plants can survive before they completely expire. Vegetation surveys during the El Niño year of 1997, when conditions were abnormally dry and many of the defoliated trees in "die-back" areas produced leaves indicate that at least some trees may survive prolonged flooding for a number of years. Long-term effects on other species are even less well understood.

Data used to develop site-specific threshold levels for stressor effects are based on field observations. Such data are important, but incomplete. However, because the screening phase looked at any adverse effects, irrespective of ecological significance, it is probably sufficiently conservative.

The natural variability in the physical processes defined as stressors and the overlap in exposure areas for these stressors make it difficult to define the normal range for these parameters in order to determine the incremental contribution due to mine waste disposal. Existing data from the study area and published information on the general range of parameters made it possible to develop threshold criteria for floodplain aggradation and flooding; however, it was not possible to quantify the incremental increase in scouring.

Threshold values for flooding frequency and floodplain aggradation were compared to predicted values for these parameters developed by Klohn-Crippen (1996). If any predicted values exceeded effects thresholds for the null case (no mitigation), for a point within a particular river reach at any time within the 50+ years the model evaluated, that reach was identified for further consideration in the DLRA. Floodplain levels were read directly from figures illustrating floodplain aggradation. Flooding frequency was read as percent exceedance at floodplain level rather than bank level. The rationale for choosing this parameter is based on the conservative assumption that riverbanks were higher than floodplain level and that these would be breached by increased flooding. While the model parameters used to predict floodplain aggradation and flooding were conservative, inaccuracies in some input values (e.g., floodplain width and elevation) may have resulted in errors and corresponding uncertainties in estimates for the middle Fly River. Sediment depth predictions based on these input parameters suggest that this is true.

In summary, the identified uncertainties were acceptable for evaluating physical stressors and effects on terrestrial vegetation in the SLRA. Additional data to reduce the uncertainties associated with threshold effects, aerial extent of effects, severity, and duration of exposure and effects will be needed for the quantification of risks in the DLRA.

5.5.2.2 Chemical Stressors

Primary data gaps identified from the SLRA included the following.

1) Floodplain sediment data were available for copper from Reaches 3, 4, and 5. At Reach 5, with the exception of copper, only within-river and ORWB sediment data were available. There were no floodplain sediment data available from Reach 6. Only within-river sediment data were available from the estuary;

2) There were no sediment data available for selenium. As such, no assessment of selenium could be undertaken;

3) There are limited published phytotoxicity data available with which to derive phytotoxicity benchmarks. There are less data available for endemic PNG plant species. Consequently, confidence in the appropriateness of the phytotoxicity benchmarks used in this assessment is low. The phytotoxicity benchmarks are considered to be protective of most plant species as they typically have been derived from sensitive agricultural crop species (i.e., tomatoes, lettuce) and effects have been evaluated using sensitive endpoints (e.g., root elongation, growth, crop yield). No phytotoxicity-based soil benchmark was available for iron;

4) The soil phytotoxicity-based soil benchmarks used in this assessment have typically been derived from plant toxicity tests using terrestrial plant species grown in soils, and the phytotoxicity benchmarks may not be appropriate for use when assessing emergent vegetation growing in sediments that can tolerate periodic waterlogging, such as the vegetation communities that occur in Reach 3 and further downstream. Thus, it has been assumed that the proposed phytotoxicity-based soil benchmarks are protective of all plant forms (terrestrial and semi-aquatic). Mangroves and other emergent plant forms may potentially be more tolerant of metals than terrestrial plant species grown in soils, and the tolerance of certain aquatic plants (e.g., *Typha* sp.) to SOPCs has been adapted to effluent treatment in artificially constructed wetlands;

5) The phytotoxicity of aluminium is dependent on the soil pH in which the plant is exposed. Under neutral or alkaline soil condition, no phytotoxicity would be likely. At pH 4, aluminium may express adverse effects on exposed plants. Limited pH data were available, however, it is understood that the pH is stable at pH 8.3 in the Ok Tedi River and 7.8 at Obo. Under these conditions, it is unlikely that aluminium would pose an unacceptable risk to vegetation communities along the river system;

6) Information on the regrowth of plants on mine-derived sediments was not available to comment on whether the sediments are being colonised in the field naturally;

7) There were no sediment data from Reach 1. Due to the high flow rate of water in the river at Reach 1, it is unlikely that SOPCs in sediments would deposit in this area. Thus, it is unlikely that SOPCs in sediments would pose a risk to terrestrial plants in Reach 1. Sediment data from Reach 2 have been assumed to represent typical SOPC concentrations in Reach 1;

8) Sediment data were available for a range of SOPCs from Reach 2 and Reach 5. The data allowed some extrapolation to the floodplain sediment quality in the reaches between, however, actual sediment data from these reaches would have provided for a more accurate assessment; and

9) From this screening-level assessment, it has not been possible to draw definitive conclusions on the factor(s) producing the dieback and stressed vegetation observed along the river system in Reaches 2, 3, and 4. Flooding, waterlogged soils, sedimentation and exposure to elevated SOPC concentrations in the mine-derived sediments are potential stressors in these areas. However, as indicated above, unless considerable leaching of SOPCs occurs in the soils, they are unlikely to contact deep-rooted plant species, such as trees, and thus elevated SOPC concentrations in the sediments may not be a cause of the dieback and stress observed in established trees. Detailed survey information on the incidence of dieback and stressed vegetation in areas affected and unaffected by mine-derived sediments was not available. However, the area of dieback is substantially larger than the area of mine-derived sediment deposition. Much of this dieback is apparently the result of increased flooding resulting from sediment deposition in in the river channel. Thus, it is unlikely that mine-derived sediment is causing the dieback observed.

5.5.3 Aquatic Life

As in the human health and terrestrial risk assessments, the primary uncertainties in the aquatic life SLRA are the data gaps in both sediment and surface water. For some metals and metalloids (e.g., mercury and selenium), sediment and surface water data are not available for any of the reaches evaluated. For others, such as chromium, lead, and nickel, limited data are available in only one or two reaches. Copper was the only analyte extensively sampled in both sediment and surface water in most (but still not all) reaches. Table 33 shows the analytes that have been measured in each reach, matrix (sediment and surface water), and site type (e.g., main stem, ORWB).

In addition to the data gaps associated with dissolved metal concentrations, no assessment has been made regarding the potential effects of particulate metals (particularly copper) to aquatic life. Considering the significant metal loads associated with the particulate phase, this creates considerable uncertainty in the overall aquatic risk assessment.

The sediment and surface water criteria are another source of uncertainty. None of the criteria were derived based on toxicity data for PNG species. Consequently, it is unknown if the criteria are over- or under-protective of PNG aquatic life. However, given that the criteria are designed to be conservative, it is more likely that they are over-protective than under-protective.

The most significant uncertainties in the sediment hazard quotients are the (1) the lack of background sediment concentrations and (2) the lack of information on bioavailability (e.g., AVS:SEM). Without these data it is not possible to determine if the sediment HQs are elevated due to metals releases from mining activities or if the bioavailability of the metals is being overestimated.

Finally, no assessment has been made at this point regarding pH (i.e., ARD) as a potential stressor.

Table 33. Available data for metals/metalloids in sediment and surface water.

Reach	Site Type	Sediment	Surface Water (dissolved)
1	Main Stem	None	Cd, Cu, Fe, Mn, Zn
2	Main Stem	Al, As, Cd, Cr, Cu, Fe, Pb, Mn, Mo, Ni, Ag, Zn	Cd, Cu, Fe, Pb, Mn, Zn
3	Main Stem	None	Cd, Cu, Fe, Mn, Zn
	Main Stem/ORWB Ref.		Cu, Fe, Mn, Zn
4	Main Stem		Cu
	ORWB	Cu	Cu, Fe, Mn, Zn
	Floodplain	Cu	
5	Main Stem, ORWB	Al, As, Cd, Cr, Cu, Fe, Pb, Mn, Mo, Ni, Ag, Zn	Cd, Cu, Fe, Mn, Zn
	Floodplain	Cu	Cd, Cu, Fe, Mn, Zn
	Main Stem Ref.		Cu, Fe, Mn, Zn
6	Main Stem	None	Cd, Cu, Fe, Mn, Zn
Estuary		Cu	None

5.5.4 **Wildlife**

The uncertainties for wildlife receptors are related to the dose estimates and estimates of chemical concentrations in prey. For example, the results for the white-headed stilt are very conservative since it was assumed that 100 percent of the metal in the ingested food items and incidental sediment ingestion was bioavailable. This is in keeping with a conservative screening level risk assessment approach (U.S. EPA 1997). However, metals are typically much less bioavailable from food items (Hunter and Johnson 1982; Pascoe et al., 1994). Furthermore, there is uncertainty in the estimates of chemical concentrations in prey items for the whited-headed stilt and fruit bat based on BCFs and BAFs. These estimates are usually much more conservative than actual measured concentrations in prey because, BCFs and BAFs do not account for factors that affect the bioavailability of metals to prey items (e.g., hardness in water and cation exchange capacity in soil). Finally, in some cases the ingestion of food was not evaluated for a receptor due to inadequate data. For example, no estimates of the ingestion of lead in food were evaluated for the white-headed stilt in Reach 6 (main stem).

There also are uncertainties in the toxicity data used in this assessment. Specifically, we do not know if the surrogate species are as sensitive or less sensitive than the receptor. This uncertainty exists for all metals evaluated.

Finally, there are two key uncertainties in the risk estimates for terrestrial invertebrates, (1) are terrestrial invertebrates in tropical ecosystems as sensitive to metals as those in used to develop toxicity thresholds and (2) by assuming that metals in sediment would be the same as that in soil exposure levels may have been overestimated.

6. RECOMMENDATIONS

This section contains recommendations on data needs to further reduce uncertainty in the SLRA. The SLRA screened out stressors, receptors, and areas that did not require further evaluation. Terrestrial vegetation and aquatic life were carried forward into the DLRA. The following recommendations are limited to the human health and wildlife components of the risk assessment as these components are not carried forward to the DLRA. Recommendations for terrestrial vegetation and aquatic life are presented in the DLRA report after completion of this assessment.

6.1 HUMAN HEALTH

Recommendations on how to address uncertainties identified in the human health and wildlife SLRAs have been developed during this study (recommendations for terrestrial vegetation and aquatic life are provided in the DLRA). Recommendations and data needs are summarised below; they relate to additional studies, characterisation of environmental media (e.g., soil, sediment), and resource use by local peoples.

6.1.1 Characterisation of Environmental Media

Additional information is needed to characterise SOPC concentrations in many of the environmental media to which humans may be exposed. Sediment data were used in the SLRA to characterise exposures that could occur along river banks or in the floodplain of the Fly River. Similar data for soils that may be affected by deposition of sediments containing mine waste was not available. Soil data collected from gardens or other areas where contact may occur would allow the evaluation of similar exposures for this medium. Soil data should be collected at locations where contact may occur (e.g., child play areas, gardens), areas that may be affected by mine waste, and background (i.e., non-mine-impacted) areas.

Additional data on chemical concentrations in food items from locations that are customary for fishing and gathering by local people are needed. Specifically, data are needed for the fish, invertebrate and plant species that are known to be consumed by local people. The extent to which the currently available fish tissue data reflect species that are consumed by local people is uncertain. It is also important to document the locations of customary fishing and gathering areas relative to tailings-affected media so it can be established whether there is a link to mine waste. Fish tissue data collected should include both the edible portion of the fish (one subsample) and also the remainder of the fish (second subsample) for each key species consumed. Fish tissue data should be collected from reference areas to compare risks from areas unaffected by mine waste.

6.1.2 Resource Use by Local People

The intensity with which aquatic food resources are used by local people (i.e., ingestion rates) were derived from literature values in the SLRA. A site-specific survey to characterise the amount (mass), frequency (days/year) and species consumed in the Ok Tedi region (Reach 2 and 3) are needed to better characterise exposure to chemical stressors in aquatic food resources.

Additionally, better information to characterise sources of drinking water is needed to ensure that drinking water resources are not affected by mine waste. Further, site-specific body weight data could be collected during this survey to allow a more accurate estimate of chemical intakes.

6.1.3 Other Studies

Additional studies could be conducted, if warranted by analysis of data collected in the recommendations above, to determine if levels of chemicals in human tissue are elevated (i.e., biomarker studies). In general, biological levels are not well correlated well with effects (e.g., cadmium in hair). However, biomarkers are available to indicate if excessive exposures may be occurring. For lead, concentrations in blood are a widely used marker of recent lead exposures (ATSDR 1997). While levels of lead in blood may be used to indicate if exposure is occurring, they cannot (alone) be used to indicate if exposures are attributable to mine waste exposure or another source (e.g., smoking, use of leaded gasolines). Therefore, if biomarker data were to be collected, similar data from villages unaffected by mine waste would be needed to provide a reference for background levels in PNG. Similarly, reference data for the currently existing hair data would allow interpretation of elevated concentrations in hair compared to background concentrations.

6.2 WILDLIFE

As discussed in Section 5.5.4, the key uncertainties in the wildlife SLRA are related to the lack of sufficient data for certain environmental media (e.g., soil, tissue). Other uncertainties stem from the lack of understanding on the bioavailability of metals to wildlife. Recommendations on how to address these uncertainties are described below.

6.2.1 Chemistry Data

Given the importance of food items (e.g., fish, fruit) and soil as sources of chemical exposures for wildlife, it is recommended that chemistry data be collected for these media. Currently, chemical concentrations in fish, aquatic invertebrates, and terrestrial vegetation are being estimated from either surface water or sediment concentrations using BCFs or BAFs from the scientific literature. These estimated tissue concentrations are highly uncertain because of the many site-specific factors that can influence uptake. Analysis of metals in fruit and aquatic invertebrates in particular would help reduce some of the uncertainties in the risk estimates for the fruit bat and stilt. It is recommended that these data be collected in all relevant reaches.

Due to the lack of soil data, sediment data were used to estimate chemical concentrations in soil. Soil concentrations are used to estimate concentrations in food items (although, as discussed above, it is recommended that actual tissue data be collected) and to evaluate exposure of terrestrial invertebrates. Given the high HQs for terrestrial invertebrates, these data would provide more realistic exposure estimates.

6.2.2 Bioavailability

In addition to collecting soil chemistry data, earthworm bioassays would provide information on the bioavailability of metals in site soil. The high HQs that have been calculated for terrestrial invertebrates are based on conservative, non-site-specific toxicity values. It is recommended that bioassays are conducted using soil collected from several locations along each river reach. This would ensure that within-site variability in metals concentrations and bioavailability is adequately assessed. Soils from all reaches should be tested in order to reflect differences in metals concentrations and bioavailability as different types of sediment are deposited from upstream to downstream. As discussed above, metals concentrations should be measured in the soil samples in order to determine the metals potentially associated with any observed toxicity.

7. REFERENCES

Abernathy, C.O. and S.M. Roberts. 1998. Possible Revisions to the Arsenic Human Health Risk Assessment. Office for Science and Technology, U.S. Environmental Protection Agency, Unpublished Manuscript.

Adema, D.M.M. and Henzen, L. 1989. A comparison of plant toxicities of some industrial chemicals in soil culture and soilless culture. Ecotox. Environ. Saf. 18:219-229.

Adriano, D. C. 1986. Trace elements in the terrestrial environment. New York: Springer Verlag.

Ahlrichs, J.L., Karr, M.C., Baligar, V.C. and Wright, R.J. 1990. Rapid bioassay of aluminium toxicity in soil. Plant and Soil. 122: pp. 279-285.

Alcoa. 1996. Volume B2b: Bay System Investigation Phase 2A. Prepared by Parametrix, Inc., Kirkland, WA.

Alloway B. J. and A.P. Jackson. 1991. The behaviour of heavy metals in sewage sludge-amended soils. Sci. Total Environ. 100:151-176.

Alloway, B. J. 1990. Soil processes and the behaviour of metals. *In*: Heavy metals in soils, ed B.J. Alloway. Glasgow: Blackie and Son Ltd.

Alloway, B.J. 1995. Heavy Metals in Soils. 2nd Ed. Blackie Academic and Professional.

Ambrose, A.M., P.S. Larson, J.F. Borzelleca, and G.R. Hennigar, Jr. 1976. Long-term toxicologic assessment of nickel in rats and dogs. J. Food Sci. Tech. 13:181-187.

American Cancer Society. 1993. Cancer facts and figures – 1993. American Cancer Society, Atlanta Georgia.

Anderson, R. and Christensen, T. H. 1988. J. Soil Science. 39: 15-22. In: Alloway 1995, page 166.

Andrew, R.W. 1976. Toxicity relationships to copper forms in natural waters. Pages 127-143. R.W. Andrew, P.V. Hodson, and D.E. Konasevich, editors. Toxicity to biota of metal forms in natural water. Proceedings of a workshop held in Duluth, Minnesota, October 7-8, 1975. International Joint Commission, Windsor, Ontario.

Ankley, G.T., G.L. Phipps, E.N. Leonard, D.A. Benoit, and U.P. Mattson. 1991. Acid-volatile sulfide as a factor mediating cadmium and nickel bioavailability in contaminated sediments. Environ. Toxicol. Chem. 10:1299-1307.

ANZECC (Australian and New Zealand Environmental Conservation Council). 1992. Australian water quality guidelines for fresh and marine waters. National Water Quality Management Strategy. Australian and New Zealand Environment and Conservation Council.

ANZECC (Australian and New Zealand Environmental Conservation Council). 1997. ANZECC Interim Sediment Quality Guidelines. Prepared for Environmental Research Institute of the Supervising Scientist. March 1997 (draft).

ANZECC (Australian and New Zealand Environmental Conservation Council). 1997. ANZECC Interim Sediment Quality Guidelines. Prepared for Environmental Research Institute of the Supervising Scientist. March 1997 (draft).

ATSDR (Agency for Toxic Substances and Disease Registry). 1999. Toxicological Profile for Mercury. Agency for Toxic Substances and Disease Registry, U.S. Public Health Service. Prepared by: Research Triangle Institute under contract No. 205-93-0606. March, 1999.

Aulerich, R.J., R.K. Ringer, M.R. Bleavins, et al., 1982. Effects of supplemental dietary copper on growth, reproductive performance and kit survival of standard dark mink and the acute toxicity of copper to mink. J. Animal Sci. 55:337-343.

Barghigiani, C. and T. Ristori. 1995. Preliminary study on mercury uptake by *Rosmarinus officinalis* L. (rosemary) in a mining area (Mt. Amiata, Italy). Bull. Environ. Contam. Toxicol. 54:519-525.

Barnthouse, L.W., G.W. Suter II, S.M. Bartell, J.J. Beauchamp, R.H. Gardner, E. Linder, R.V. O'Neill, and A.E. Rosen. 1986. User's manual for ecological risk assessment. Environmental Sciences Division Publication No. 2679. Prepared for U.S. EPA, Office of Research and Development. Report No. ORNL-6251. 215 pp.

Bartlett, R.J. 1986. Chromium oxidation in soils and water: measurements and mechanisms. *In*: Proceedings Chromium Symposium: An update, ed. D,M. Serrone. Industrial Health Foundation.

Bartlett, R.J. 1991. Chromium cycling in soils and water: Links, gaps, and methods. Environ. Health Perspec. 92:17-24.

Bartlett, R.J. and B.R. James. 1979. Behaviour of chromium in soils: III. Oxidation. J. Environ. Qual. 8:31-35.

Bartlett, R.J. and B.R. James. 1988. Mobility and bioavailability of chromium in soils. *In*: Chromium in the natural and human environments, ed. J.0. Nriagu, and E. Nieboer. Toronto: John Wiley and Sons.

Bartlett, R.J. and J.M. Kimble. 1976. Behaviour of chromium in soils: Hexavalent forms. J. Environ. Qual. 5:383-386.

Bearer, C.F. 1995. How are children different from adults. Environmental Health Perspectives. 103(Suppl.6): 7-12.

Beeby, A. 1991. Toxic metal uptake and essential metal regulation in terrestrial invertebrates: A Review. *In*: Metal Ecotoxicology: Concepts and Applications. M.C. Newman and A.W. McIntosh, eds. Lewis Publishers, Chelsea, Michigan.

Beehler, B. M., T. K. Pratt, and D. A. Zimmerman. 1986. Birds of New Guinea. Princeton University Press, Princeton.

Begearmi, M.M., H.E. Ganther, and M.L. Sunde. 1980. Toxicity of mercuric chloride in Japanese quail as affected by methods of incorporation into the diet. Poult. Sci. 59(10):216-2220.

Benjamin, M.M. and J.O. Leckie. 1981a. Competitive adsorption of Cd. Cu, Zn and Pb on amorphous iron oxyhydroxide. J. Colloid Interface Sci. 83:410-419.

Benjamin, M.M. and J.O. Leckie. 1981b. Multiple-site adsorption of Cd, Cu, Zn and Pb on amorphous iron oxyhydroxide. J. Colloid Interface Sci. 79:209-221.

Beyer, W.N., O.H. Pattee, L. Sileo, D.J. Hoffman, and B.M. Mulhern. 1985. Metal contamination in wildlife living near two zinc smelters. Environmental Pollution (Series A). 38:63-86.

Biddinger, G.R, and Gloss, S. 1984. The importance of trophic transfer in the bioaccumulation of chemical contaminants in aquatic ecosystems. Pages 104-145. *In*: Residue Reviews. Springer-Verlag, New York, New York. Volume 91.

Bigham N.G., and A.E. Henry. 1993. Mercury in sediments - How clean is clean? *In*: Mercury and arsenic wastes. U.S. EPA. Pollution Technology Review No.214:11-13.

Blackwood, B. 1940. Use of Plants among the Kukukuku of Southeast Central New Guinea Proceedings of the Sixth Pacific Science Congress. Vol. 4:11-126.

Block, M. and A. Wicklund Glynn. 1992. Influence of xanthates on the uptake of Cd by Eurasion dace (*Phoxinus phoxinus*) and rainbow trout (*Oncorhynchus mykiss*). Environ. Toxicol. Chem. 11: 873.

Bloomfield, C. and W.I. Kelso. 1973. J. Soil. Sci. 24: pp. 368. (*In*: Alloway 1995, page 320).

Bloomfield, C. and G. Pruden. 1980. The behaviour of Cr(VI) in soil under aerobic and anaerobic conditions. Environ. Pollut. Series A. 23:103-114.

Blume, H.P. and G. Brummer. 1991. Prediction of heavy metal behaviour in soil by means of simple field tests. Ecotoxicol. Environ. Saf. 22:164-174.

Bodek, I. 1988. Environmental inorganic chemistry: properties, processes, and estimation methods. Pergamon Press, Elmsford, New York.

Bollag, J. M and J. Czaban. 1989. Effect of micro-organisms on extractability of cadmium from soil with sodium hydroxide and DTPA. J. Soil Sci. 40:451-460.

Bolton, M. 1976. Crocodile farming in Papua New Guinea. Ethnology. Vol. 15(3):365-369.

Borg, K., J. Gottofrey, and H. Tjalve. 1988. Effects of some chelating agents on the uptake and distribution of Hg (II) in the brown trout (*Salmo trutta*): studies on ethyl – and isopropylxanthate, diethyl- and diisopropyldithiophosphate, dimethyl- and diethyldithiocarbamate and pyridinethione. Arch. Toxicol. 62: 387.

Bou-Olyan, A.H. and S.N. Al-Yakoob. 1994. Mercury in human hair: A study of residents of Kuwait. Journal of Environmental Science and Health. A29(8):1541-1551.

Bourke, R. M. 1990. Subsistence food production systems in Papua New Guinea: Old Changes and New Changes. *In*: D .E. Yen and J. M. J. Mummery eds., op.cit.

Bourliere, F. 1983a. Animal species diversity in tropical forests. Chapter 5 *In*: F. B. Golley ed. Ecosystems of the World 14A: Tropical Rain Forest Ecosystems. Elsevier Scientific Publishing Co., Amsterdam.

Bourliere, F. 1983b. Savannas of Australia and Southwest Pacific. *In:* (Bourliere, F. (ed.) Ecosystems of the World 13: Tropical Savannas. Elsevier Scientific Publishing Company, Amsterdam.

Bowen, H. J. M. 1966. Trace elements in biochemistry. Academic Press, London. 241 pp.

Brewer, R. 1979. Principles of ecology. W. B. Saunders and Company, Toronto.

Bright, C. 1999. The nemesis effect. World Watch. 12(3):12-19.

Brookes, R.R. 1987. Serpentine and Its Vegetation. Croom Helm, London.

Brown, K.G. and C.O Abernathy. 1997. The Taiwan skin cancer risk analysis of inorganic arsenic ingestion: effects of water consumption rates and food arsenic levels. Pages 260-271. *In*: Arsenic Exposure and Health Effects, CO Abernathy, RL Calderon, and WR Chappell, eds. Chapman & Hall, New York.

Brown, J. C. 1976. Screening of plants for iron efficiency. In: M.J. Wright, ed. Plant Adaptation to Mineral Stress in Problem Soils. Cornall University Agricultural Experimental Station, Ithaca. pp. 355-358.

Brown, K.G. 1998. Assessing Risk of Inorganic Arsenic in Drinking Water in the United States. Unpublished Manuscript.

Bruemmer, G.W., J. Gerth, and K.G. Tiller. 1988. Reaction kinetics of the adsorption and desorption of nickel, zinc and cadmium by goethite. I. Adsorption and diffusion of metals. J. Soil Sci. 39:37-52.

Bull, K.R., R.D. Roberts, M.J. Inskip, and G.T. Goodman. 1977. Mercury concentrations in soil, grass, earthworms and small mammals near an industrial emission source. 12:135-140.

Bulmer, R. N. H. 1968. The Strategies of Hunting in New Guinea, Oceania 38: 302-318.

Burgin, S. 1980. A review of crocodile farming in Papua New Guinea, Science in New Guinea, 7(2): 73-88.

Burton, Jr., G.A. 1992. Sediment collection and processing: Factors affecting realism. Pages 37-66. *In:* G.A. Burton, Jr., editor. Sediment toxicity assessment. Lewis Publishers, Boca Raton, Florida. 457 pp.

Cain, B.W., and E.A. Pafford. 1981. Effects of dietary nickel on survival and growth of mallard ducklings. Arch. Environ. Contam. Toxicol. 10:737-745.

Callahan, M.A. M.W. Slimak, N.W. Gabel, I.P. May, C.F. Fowler, J.R. Freed, P. Jennings, R.L. Durfee, F.C. Whitmore, B. Maestri, M.R. Mabey, B.R. Holt, and C. Gould. 1979. Water-related environmental fate of 129 priority pollutants. U.S. EPA-440/4-79-029a,b.

Cameron, R. D. 1973. Unpubl. PhD thesis, Colorado State University. (In: Alloway 1995, pages 326-327).

Campbell, P.G. and A.G. Lewis. 1988. Biologically available metals in sediments. National Research Council of Canada, Ottawa, Canada.

Cappon, C.J. 1987. Uptake and speciation of mercury and selenium in vegetable crops grown on compost-treated soil. Water, Air, and Soil Pollution. 34:353-361.

Carlson, A.R., G.L. Phillips, V.R. Mattson, P.A. Kosian, and A.M. Cotter. 1991. The role of acid-volatile sulfides in determining cadmium bioavailability and toxicity in freshwater sediments. Environ. Toxicol. Chem. 10:1309-1319.

Carriere, D., K.L. Fischer, D.B. Peakall, and P. Anghern. 1986. Effects of dietary aluminium sulfate on reproductive success and growth of ring doves. Can. J. Zool. 64:1500-5.

Carroll, B.I., J.P. Barford, K. Gotham, and M. Reso. 1996. Selenium transformations and metabolism in bacterial isolates from Lake Macquarie NSW. 23rd Hydrology and water Resources Symposium, 21-24 May, 1996, Hobart, Australia.

Casas, G.A. and E.A. Crecelius. 1994. Relationships between acid-volatile sulfide and the toxicity of zinc, lead, and copper in marine sediments. Environ. Toxicol. Chem. 13:529-536.

Cassarett and Doulls. 1996. Toxicology: The basic science of poisons. 5th. Ed. Macmillan Publishing Company. Klassen, C.D. ed. New York 1996.

CCME (Canadian Council of Ministers of the Environment). 1994. A framework for ecological risk assessment: general audience. Final Report. Evaluation and Interpretation Branch, Ecosystem Conservation Directorate, Environment Canada, Ottawa, Canada.

CCME (Canadian Council of Ministers of the Environment). 1995. Canadian water quality quidelines. Prepared by the Task Force on Water Quality Guidelines of the Canadian Council of Ministers of the Environment. Environmental Quality Guidelines Division, Ottawa, Ontario.

CCME (Canadian Council of Ministers of the Environment). 1996. Canadian Soil Quality Guidelines for Copper: Environmental and human health. Draft document.

CCME (Canadian Council of Ministers of the Environment). 1997a. Canadian Soil Quality Guidelines for Copper; Environmental and Human Health. Prepared by the CCME Subcommittee on Environmental Quality Criteria for Contaminated Sites.

CCME (Canadian Council of Ministers of the Environment). 1997b. Recommended Canadian Soil Quality Guidelines. March 1997. CCME, Guidelines Division, Science Policy and Environmental Quality Branch, Ottawa, Ontario.

Chandra, P. and P. Garg. 1992. Absorption and toxicity of chromium and cadmium in *Limnanthemum cristatum* Griseb. The Science of the Total Environment, 125, 175-183.

Chanmugathas, P., and J.M. Bollag. 1987. Microbial mobilisation of cadmium in soil under aerobic and anaerobic conditions. J. Environ. Qual. 16:161-167.

Chapman, W.H., H.L. Fisher, and M.W. Pratt. 1968. Concentration factors of chemical elements in edible organisms. University of California, Lawrence Radiation Laboratory, Bio-medical Division. Livermore, Calif. UCRL-50564. 46p.

Chapman, P.M. 1996. Hazard identification hazard classification and risk assessment for metals and metal compounds in the aquatic environment. International Council on Metals and the Environment. Ottawa, Canada.

Chigbo, F.E., R.W. Smith, and F.L. Shore. 1982. Uptake of arsenic, cadmium, lead and mercury from polluted waters by the water hyacinth *Eichhornia crassipes*. Environ. Pollut. Ser. A. 0143-1471, 31-36.

Christensen, T.H. 1984a. Cadmium soil sorption at low concentrations: I. Effect of time, cadmium load, pH, and calcium. Water Air Soil Pollut. 21:105-114.

Christensen, T.H. 1984b. Cadmium soil sorption at low concentrations: II. Reversibility, effect of changes in solute composition, and effect of soil ageing. Water Air Soil Pollut. 21:115-125.

Christensen, T.H. 1989. Cadmium soil sorption at low concentrations. VIII. Correlation with soil parameters. Water Air Soil Pollut. 44: 71-82.

Cogger, H.G. 1994. Reptiles and amphibians of Australia. Reed Books and Cornell University Press. Ithaca, New York.

Collins, J.C. 1981. Nickel. *In:* Effects of heavy metal pollution on plants, Vol. 1. Effects of Trace Metals on Plant Function. Chapter 6. Applied Science Publishers, NJ. pp. 145-170.

Compeau, G., and R. Bartha. 1984. Methylation and demethylation of mercury under controlled redox, pH, and salinity conditions. App. Environ. Microbio. 48:1203-1207.

Cooper, C. F. and Jolly, W. C. 1970. Ecological effects of silver iodide and other weather modification agents: A Review. Water Resources Res. 6: 88-98

Crawford, L.A., I.D. Hodkinson, and N.W. Lepp. 1995. The effects of elevated host-plant cadmium and copper on the performance of the aphid Aphis fabae (Homoptera: Aphidae). J. Appl. Ecol. 32:528-535.

Cross, S.J. and Taylor, E.R. 1996. Human exposure to soil contaminants through consumption of home-grown produce. Contaminated Sites Monograph Series No. 6.

CSIRO. 1996. Review of riverine impacts: Porgera Joint Venture. CSIRO Environmental Projects Office, Australia.

CSIRO. 1997. Bioavailability of dissolved copper in waters downstream of the Ok Tedi mine. Prepared for Ok Tedi Mining Limited. CSIRO, Bangor, Australia. December 1997. 58 pp.

Cui, Y. and G. Parker. 1999. Sediment transport and deposition in the Ok Tedi-Fly River System, Papua New Guinea: The Modeling of 1998-1999. St. Anthony Falls Laboratory, University of Minnesota.

Cureton, P.M., Balch, G., Lintott, D. Poirrier, K., Gaudet, C. and Goudey, S. 1994. Toxicity testing of NCSRP priority substances for the development of soil quality criteria. Page 342. Vancouver, BC. SETAC Abstract.

D'Itri, F.M. 1990. The biomethylation and cycling of selected metals and metalloids in aquatic systems. In: Sediments: Chemistry and toxicity on in-place pollutants. R. Bando, J. Giesy, and H. Muntau, eds. Lewis Publishers, Chelsea, Michigan.

D'Itri, F.M. 1993. Mercury pollution and cycling in aquatic systems. Environmental Contamination, 5th International Conference, September 29-October 1, 1992. Pg. 391-402.

Dallinger, R., F. Post, H. Segner, and H. Back. 1987. Contaminated food and uptake of heavy metals by fish: a review and proposal for further research. Oecologia, 73:91-98.

Darlington, P. J., Jr. 1957. Zoogeography: the geographic distribution of animals. John Wiley and Sons, Inc.

Davidson, P.W., G.J. Myers, and C. Cox. 1998. Effects of prenatal and postnatal methylmercury exposure from fish consumption on neurodevelopment: Outcomes at 66 months of age in the seychelles child development study. JAMA 280(8):701-707.

Davies, P.H. 1976. The need to establish heavy metal standards on the basis of dissolved metals. Pages 93-126. *In:* R.W. Andrew, P.V. Hodson, and D.E. Konasewich, editors. Toxicity to biota of metal forms in natural water. Proceedings of a workshop held in Duluth, Minnesota, October 7-8, 1975.

Davies, P.H., J.P. Goettl, Jr., J.R. Sinley, and N.F. Smith. 1976. Acute and chronic toxicity of lead to rainbow trout Salmo gairdneri in hard and soft water. Water Research. 10:199-206.

Davis-Carter, J.G. and L.M. Schuman. 1993. Influence of texture and pH of kaolinitic soils on zinc fractions and zinc uptake by peanuts. Soil Sci. 155:377-385.

Di Toro, D.M., S.M. Mahond, D.J. Hansen, K.J. Scott, A.R. Carlson, and G.T. Ankley. 1990. Toxicity of cadmium in sediments: the role of acid-volatile sulfide. Environ. Toxicol. Chem. 9:1487-1502.

Di Toro, D.M., S.M. Mahond, D.J. Hansen, K.J. Scott, A.R. Carlson, and G.T. Ankley. 1992. Acid-volatile sulfide predicts the toxicity of cadmium and nickel in sediments. Environ. Toxicol. Chem. 26:96-101.

Dickens, R. and A.E. Hiltbold. 1967. Movement and persistence of methanearsonates in soil. Weeds 15:299-304.

Dieter, M.P., M.I. Luster, G.A. Boorman, C.W. Jameson, J.H. Dean, and J.W. Cox. 1983. Immunological and biochemical responses in mice treated with mercuric chloride. Toxicol. Appl. Pharmacol. 68: 218-228.

Dourson, M.L., S.P. Felter, and D. Robinson. 1996. Evolution of science-based uncertainty factors in non-cancer risk assessment. Regulatory Toxicol and Pharmacol: 24:108-120.

Duff, G. A. 1992. Effects of sediment deposits on rainforest trees on the Ok Tedi – Final Report to Ok Tedi Mining Limited. Northern Territory University.

Dunning, Jr., J.B. 1993. Avian body masses. CRC Press, Boca Raton, Florida. 371pp.

Duquette, M. and W.H. Hendershot. 1990. Copper and zinc sorption on some B horizons of Quebec soils. Commun. Soil Sci. Plant Anal. 21:377-394.

Eary, L.E. and D. Rai. 1991. Chromate reduction in subsurface soils under acidic conditions. Soil Sci. Soc. Am. J. 55:676-683.

Edens, F., W.E. Benton, S.J. Bursian, and G.W. Morgan. 1976. Effect of dietary lead on reproductive performance in Japanese quail, Coturnix coturnix japonica. Toxicol. Appl. Pharmacol. 38:307-314.

Edwards, R., N.W. Lepp, and K.C. Jones. 1995. Chapter 14: Other less abundant elements and potential environmental significance. *In*: Alloway, B. J. ed.Heavy Metals in Soils. Blackie Academic & Professional Publishers.

Efroymson, R.A., M.E.Will, G.W. Suter II, and A.C. Wooten. 1997. Toxicological benchmarks for screening contaminants of potential concern for effects on terrestrial plants: 1997 *Revision*. Prepared for the US Department of Energy, Office of Environmental Management.

EIFAC (European Inland Fisheries Advisory Commission). 1965. Water quality criteria for European fresh water fish. Report on finely divided solids and inland fisheries.. Air Water Pollut. 9:151-168.

Eisler, R. 1985. Selenium hazards to fish, wildlife, and invertebrates: A synoptic review. U.S. Fish and Wildlife Service Biological Report No. 85:1:10.

Eisler, R. 1988. Arsenic hazards to fish, wildlife, and invertebrates: A synoptic review. U.S. Fish and Wildlife Service. Biological Report 85(1.12). 92 pp.

Eisler, R. 1993. Zinc hazards to fish, wildlife, and invertebrates: A synoptic review. U.S. Fish and Wildlife Service. Biological Report 10. 106 pp.

Eisler, R. 1997. Copper hazards to fish, wildlife, and invertebrates: A synoptic review. U.S. Geological Survey, Biological Resources Division. Biological Science Report USGS/BRD/BSR-1997-0002. 98 pp.

Elsokkary, I.H. and J.Lag. 1978. Distribution of different fractions of Cd, Pb, Zn and Cu in industrially polluted and non-polluted soils of Odda region, Norway. Acta Agric. Scand. 23: 262-268

Environment Australia. 1997. (draft) National framework for ecological risk assessment of contaminated sites. Department of Environment, Environment Australia.

Environment Canada. 1995. Toxicity testing of national contaminated sites remediation program priority substances for the Development of Soil Quality Guidelines for Contaminated Sites. Guidelines Division, Evaluation and Interpretation Branch, Environmental Conservation Service, Environment Canada, Ottawa. Unpublished.

EPA (U.S. Environmental Protection Agency). 1980. Ambient aquatic life water quality criteria for silver. Office of Water Regulations and Standards Criteria and Standards Division. United States Environmental Protection Agency, Washington, D.C. EPA 440/5-80-071.

EPA (U.S. Environmental Protection Agency). 1981. An Exposure and Risk Assessment for Silver. EPA-440/4-81-017. 160 pp.

EPA (U.S. Environmental Protection Agency). 1984. Health effects assessment for iron (and compounds). Office of Research and Development. United States Environmental Protection Agency, Cincinnati, Ohio. EPA/540/1-86-054.

EPA (U.S. Environmental Protection Agency). 1985a. Ambient water quality criteria for copper. Office of Water Regulations and Standard Criteria Standards Division, Washington, D.C. EPA 440/5-85-032, PB85-227023.

EPA (U.S. Environmental Protection Agency). 1985b. Ambient aquatic life water quality criteria for cadmium. Office of Water Regulations and Standards Criteria and Standards Division. United States Environmental Protection Agency, Washington, D.C. EPA 440/5-84-032.

EPA (U.S. Environmental Protection Agency). 1985c. Ambient water quality criteria for arsenic. Office of Water Regulations and Standard Criteria Standards Division, Washington, D.C. PB85-227445.

EPA (U.S. Environmental Protection Agency). 1985d. Ambient aquatic life water quality criteria for chromium. Office of Water Regulations and Standards Criteria and Standards Division. United States Environmental Protection Agency, Washington, D.C. EPA 440/5-84-029.

EPA (U.S. Environmental Protection Agency). 1985e. Ambient aquatic life water quality criteria for lead. Office of Water Regulations and Standards Criteria and Standards Division. United States Environmental Protection Agency, Washington, D.C. EPA 440/5-84-027.

EPA (U.S. Environmental Protection Agency). 1985f. Development of statistical distributions or ranges of standard factors used in exposure assessments. Office of Health and Environmental Assessment. United States Environmental Protection Agency, Washington, D.C. EPA 600/8-85/010. August 1985.

EPA (U.S. Environmental Protection Agency). 1986. Ambient aquatic life water quality criteria for nickel. Office of Water Regulations and Standards Criteria and Standards Division. United States Environmental Protection Agency, Washington, D.C. EPA 440/5-86-004.

EPA (U.S. Environmental Protection Agency). 1987a. Ambient aquatic life water quality criteria for silver. Office of Water Regulations and Standards Criteria and Standards Division. United States Environmental Protection Agency, Washington, D.C.

EPA (U.S. Environmental Protection Agency). 1987b. Ambient aquatic life water quality criteria for zinc. Office of Water Regulations and Standards Criteria and Standards Division. United States Environmental Protection Agency, Washington, D.C. EPA 440/5-87-003.

EPA (U.S. Environmental Protection Agency). 1988a. Ambient aquatic life water quality criteria for aluminium. Office of Water Regulations and Standards Criteria and Standards Division. United States Environmental Protection Agency, Washington, D.C. EPA 440/5-88-008.

EPA (U.S. Environmental Protection Agency). 1988b. Ambient water quality criteria for selenium. Office of Water Regulations and Standards Criteria and Standards Division, Washington, D.C. PB88-142237.

EPA (U.S. Environmental Protection Agency). 1989a. Risk assessment guidance for Superfund, volume I: human health evaluation manual (Part A) (interim final). Toxics Integration Branch, Office of Emergency and Remedial Response, Office of Solid Waste and Emergency Response, United States Environmental Protection Agency, Washington, D.C. EPA/540/1-89/002.

EPA (U.S. Environmental Protection Agency). 1989b. Risk assessment guidance for Superfund Volume II: Environmental Evaluation Manual, EPA/540/1-89/001.

EPA (U.S. Environmental Protection Agency). 1992a. Dermal exposure assessment: Principles and Applications. United States Environmental Protection Agency, Office of Health and Environmental Assessment, Washington D.C. EPA/600/8-91/011B.

EPA (U.S. Environmental Protection Agency). 1992b. Framework for ecological risk assessment. EPA/630/R-92/001. U.S. EPA Risk Assessment Forum, Washington, D.C.

EPA (U.S. Environmental Protection Agency). 1993. Wildlife exposure factors handbook, Volume 1 of II. U.S. Environmental Protection Agency, Office of Research and Development. EPA/600/R-93-187.

EPA (U.S. Environmental Protection Agency). 1994. Equilibrium partitioning approach to predicting metal bioavailability in sediments and the derivation of sediment quality criteria for metals. U.S. EPA, Office of Water and Office of Research and Development. EPA/822-D-94-002.

EPA (U.S. Environmental Protection Agency). 1995. Final water quality guidance for the Great Lakes system; final rule. Federal Register, Vol. 60, No.56. March 23, 1995. pp15366-15425.

EPA (U.S. Environmental Protection Agency). 1996a. The metals translator: Guidance for calculating a total recoverable permit limit from a dissolved criterion. U.S. EPA, Office of Water, Washington, D.C. EPA 823-B-96-007.

EPA (U.S. Environmental Protection Agency). 1996b. Exposure Factors Handbook. Update to the Exposure Factors handbook. EPA/600/8-89/043-May 1989 Vols. I-III, SAB Review Draft, United States Environmental Protection Agency, Office of Health and Environmental Assessment, Washington D.C. EPA/600/P-95/002Ba.

EPA (U.S. Environmental Protection Agency). 1996c. AQUIRE, AQUatic toxicity information retrieval database. Environmental Research Laboratory. United States Environmental Protection Agency, Duluth, Minnesota.

EPA (U.S. Environmental Protection Agency).. 1997. Ecological risk assessment guidance for Superfund: Process for designing and conducting ecological risk assessments. Interim Final, Solid Waste and Emergency Response, Environmental Protection Agency. EPA 540-R-97-006.

EPA (U.S. Environmental Protection Agency). 1998a. Guidelines for Ecological Risk Assessment. Federal Register May 14, 1998. Pp. 26846-26924.

EPA (U.S. Environmental Protection Agency). 1998b. Integrated risk information system (IRIS) on-line computer database. Information system updated regularly by U. S. Environmental Protection Agency, Washington, D.C. Available at: http://www.epa.gov/iris.

Eriksson, J.E. 1989. The influence of pH, soil type and time on adsorption and uptake by plants of cadmium added to the soil. Water Air Soil Pollut. 48:317-336.

Ernst, C.H. 1989. Turtles of the world. Smithsonian Institution Press, Washington, D.C.

EC (European Commission). 1996. Technical guidance document in support of commission directive 93/67/EEC on risk assessment for new notified substances and commission regulation (EC) no. 1488/94 on risk assessment of existing substances. Parts I-IV. European Commission, Brussels, Luxembourg. 739 pp.

Evans, L.J. 1989. Chemistry of metal retention by soils. Environ. Sci. Technol. 23:1046-1056.

Evans, R.D., D. Andrews, and R.J. Cornett. 1988. Chemical fractionation and bioavailability of cobalt-60 to benthic deposit-feeders. Canadian Journal of Fisheries and Aquatic Sciences. 45:228-236.

Fendorf, S.E., M. Fendorf, D.L. Sparks, and R. Gronsky. 1992. Inhibitory mechanisms of Cr(III) oxidation by δ-MnO2. J. Colloid Interface Sci. 153:37-55.

Fitzgerald, W.F., R.P. Mason, G.M. Vandal, and F. Dulac. 1994. Air-water cycling of mercury in lakes. Mercury pollution: Integration and synthesis. C.J. Watras and J.W. Huckabee, editors. Lewis Publishers, Boca Raton, Florida.

Flannery, T. F. 1995. Mammals of New Guinea. Cornell University Press, Ithaca, New York.

Fleming, R.W. and M. Alexander. 1973. Dimethylselenide and Dimethyltelluride Formation by a strain of Penicillium. Appl. Micro. 24(3): 424-429.

Flew, S. 1998. Human health baseline Study, Ok Tedi Mining Ltd. Tabubil, Papua New Guinea.

Foy, C. D., R.L. Chaney, and M.C. White. 1978. The physiology of metal toxicity in plants. Ann. Rev. Plant Physiol. 29: 511-566.

Friberg, L, G.F. Nordberg and V.B. Velimir, Eds. 1986. Handbook on the toxicology of metals, 2nd ed. Vol. II: Specific Metals. Elsevier Science Publishers, 1986.

Frodin, D.G. and J.L. Gressitt. 1982. Biological exploration of New Guinea. *In:* Gressit, J.L., ed. Biogeography and Ecology of New Guinea Volume 1 part 2: Man and his Impact on the Environment. Dr. W. Junk Publishers. The Hague-Boston-London.

Fu, G., H.E. Allen, and C.E. Cowan. 1991. Adsorption and copper by manganese oxide. Soil Sci. 152:72-81.

Fuller W.H. 1977. Movement of selected metals, asbestos, and cyanide in Soil: Applications to waste disposal problems. Prepared for Municipal Environmental Research Lab, Cincinnati, Ohio U.S. Environmental Protection Agency EPA-600/2-77-020.

Funakoshi, K., H. Watanabe, and T. Kunisaki. 1993. Feeding ecology of the northern Ryukyu fruit bat, *Pteropus dasymallus dasymallus*, in a warm-temperate region. J. Zool. Lond., 230:221-230.

Gad, S.C. 1989. Acute and chronic systemic chromium toxicity. The Science of the Total Environment. 86:149-157.

Gallopin, G. C. 1972. Structural properties of food webs. *In*: Patten, B.C., ed. Systems analysis and simulation in ecology. Academic Press, New York.

Ganong, G.W. 1983. Review of medical physiology. Lange Medical Publications. Los Altos, California.

Garate, A., I. Ramos, M. Manzanares, and J.J. Lucena. 1993. Cadmium uptake and distribution in three cultivars of *Lactuca* sp. Bull. Environ. Contam. Toxicol. 50:709-716.

Garg, P. and P. Chandra. 1993. The duckweed *Wolffia globosa* as an indicator of heavy metal pollution: sensitivity to Cr and Cd. National Botanical Research Institute Research Publication No. 401 (N.S.).

Geering, H.R., E.E. Carey, L.H.P. Jones, and W.H. Allaway. 1968. Solubility and redox criteria for the possible forms of selenium in soils. Soil Science Society. Amer. Proc. 32:35-40.

Gillison, A.N. 1983. Savannahs of Australia and Southwest Pacific. Chapter 9. In: Bourlier F. ed. Ecosystems of the World 13: Tropical Savannas. Elsevier Scientific Publishing Co., Amsterdam.

Gottofrey, J., K. Borg, S. Jasim, and H. Tjalve. 1988. Effect of potassium ethylxanthate and sodium diethyldithiocarbamate on the accumulation and disposition of nickel in the brown trout (*Salmo trutta*). Pharmacol. Toxicol. 63: 45.

Gregory, P. 1995. The birds of the Ok Tedi area. Independent Publishing, Port Moresby.

Grey, B.J., C.M. Harbicht, and G.R. Stephens. 1995. Mercury in fish from rivers and lakes in southwestern northwest territories. Northern Water Resources Studies Program (Canada), Ottawa, Ontario, 76 pp. NTIS No. MIC-95-08025WEP.

Gutknecht, J. 1981. Inorganic mercury (Hg2+) transport through lipid bilayer membrane. Journal of Membrane Biology. 61:61-68.

Haines, A. K. 1979. The Subsistence fishery of the Purari Delta. Science in New Guinea. 6: 80-95.

Hallacher, L.E., E.G. Kho, N.D. Bernard, A.M. Orcutt, W.C. Dudley, Jr., and T.M. Hammond. 1985. Distribution of arsenic in the sediments and biota of Hilo Bay, Hawaii. Pac. Sci., 29:266-273.

Hare, L., R. Carignan, and M.A. Huerta-Diaz. 1994. A field study of metal toxicity and accumulation by benthic invertebrates; implications for the acid-volatile sulfide (AVS) model. Limnol. Oceanogr. 39(7):1653-1668.

Hawley, J.R. 1972. The use, characteristics and toxicity of mine-mill reagents in the province of Ontario. Industrial Wastes Branch, Ontario Ministry of the Environment. Toronto, Ontario.

Heinz, G.H., D.J. Hoffman, A.J. Krynitsky, and D.M.G. Weller. 1987. Reproduction in mallards fed selenium. Environ. Toxicol. Chem. 6:423-433.

Henry, P.R., C.B. Ammerman, and R.C. Littell. 1992. Relative bioavailability of manganese from manganese-methionine complex and inorganic sources for ruminants. J. Dairy Sci., 75: 3473-3478.

Hodson, P.V., B.R. Blunt, and D.J. Spry. 1978. Chronic toxicity of water-borne and dietary lead to rainbow trout (*Salmo gairdneri*) in Lake Ontario water. Wat. Res. 12:869-878.

Hogg, T.J., J.W.B. Stewart, and J.R. Bettany. 1978. Influence of the chemical form of mercury on its adsorption and ability to leach through soils. J. Environ. Qual. 7(3):440-445.

Hood, R.D. 1985. Cacodylic acid: Agricultural uses, Biological effects and fate. VA Monogragh. Superintendent of Documents, U.S. Govt. Printing Office, Washington, D.C.

Huebert, D.B. and J.M. Shay. 1991. The effect of cadmium and its interaction with external calcium in the submerged aquatic macrophyte *Lemna trisulca* L. Aquatic Toxicology, 20, 57-72.

Hunter and Johnson. 1982. Food chain relationships of copper and cadmium in contaminated grassland ecosystem. Oikos, 38:108-117.

Ingersoll, C.G., P.S. Haverland, E.L. Brunson, T.J. Canfield, F.J. Dwyer, C.E. Henke, N.E. Kemble, D.R. Mount, and R.G. Fox. 1996. Calculation and evaluation of sediment effect concentrations for the amphipod (*Hyalella azteca*) and the midge (*Chironomus riparius*). J. Great Lakes Res. 22(3):602-623.

Inskeep, W.P. and J. Baham. 1983. Adsorption of Cd(II) and Cu(II) by Na-montmorillonite at low surface coverage. Soil Sci. Soc. Am. J. 47:660-665.

IUCN (World Conservation Union). 1996. 1996 IUCN Red List of Threatened Animals. IUCN, Gland, Switzerland

Jackson, R. 1998. Report on the Survey of lower Ok Tedi Villages, May-June 1998. Ok Tedi Mining LTD: Mine waste management Project-Socio-Economic Monitoring.

Jacobs, L.W., J.K. Syers, and D.R. Keeney. 1970. Arsenic sorption by soils. Soil Sci. Soc. Amer. Proc. 34:750-754.

Janus, J.A. and E.I. Krajnc, 1989. Integrated criteria document - Chromium: Effects. (Appendix to Report No. 758701001). National Institute of Public Health and Environmental Protection, Bilthoven, The Netherlands.

Janzen D. H. 1983. Food webs: who eats what, why, how, and with what effect in a tropical forest. Chapter 11. *In:* F.B. Golley, ed. Ecosystems of the World 14A: Tropical Rain Forest Ecosystems. Elsevier Scientific Publishing Co., Amsterdam.

Jaworski, J.F. 1985. Chromium update 1984: Environmental and nutritional effects of chromium. NRCC No. 23917. National Research Council of Canada, Associate Committee on Scientific Criteria for Environmental Quality, Ottawa.

JECFA(Joint FAO/WHO Expert Committee on Food Additives). 1989. Evaluation of certain food additives and contaminants. Thirty–Third Report of the Joint FAO/WHO Expert Committee on Food Additives. Technical Report Series. World Health Organisation, Geneva.

Jenner, H.A., J.P.M. Jansenn-Mommen, and J. H. Koeman. 1992. Effects of coal gasification slag as a substrate for the plant *Cyperus esculentus* and the worm *Eisenia fetida*. Ecotoxicology and Environmental Safety. 24:46-57.

Kabata-Pendias, A. and H. Pendias. 1984. Trace elements in soils and plants. Boca Raton, Florida: CRC Press Inc.

Kabata-Pendias, A. and H. Pendias. 1992. Trace elements in soils and plants, 2nd edition. London: CRC Press.

Khan, S., D. Nadan, and N.N. Khan, 1982. Environ. Pollut. (Ser. B), 4: 211-214. *In:* Alloway 1995, Ch 14.

Kimball, G. 1978. The effects of lesser known metals and one organic to fathead minnows (*Pimephales promelas*) and *Daphnia magna*. Unpublished report.

Kimbrough, D.E., Y. Cohen, A.M. Winer, L. Creeelmean, and C. Mabuni. 1999. A critical assessment of chromium in the environment. Critical Reviews in Environmental Science and Technology. 29(1)1-46.

Klohn-Crippen. 1996. Waste Management Study: Ok Tedi and Fly River Model Report (2 volumes).

Konemann, W.H. and M.N. Pieters. 1996. Confusion of concepts in mixture toxicology. Food and Chemical Toxicology, 34:1025-1031.

Kormondy, E. J. 1969. Concepts of ecology. Prentice-Hall, Inc., Englewood Cliffs, New Jersey.

Kozlowski, T. T. 1984. Flooding and plant growth. Academic Press, Orlando, Florida.

Kuffler, S.W. and J.G. Nichols. 1976. From neuron to brain. Sinauer Associates, Inc. Sunderland, Massachusetts.

Kyle, J.H. and N. Ghani. 1988. Methylmercury in human hair: A study of a Papua New Guinean population exposed to methylmercury through fish consumption. Archives of Environmental Health. 37(5): 266-270.

Laskey, J.W., G.L. Rehnberg, J.F. Hein, and S.D. Carter. 1982. Effects of chronic manganese (Mn_3O_4) exposure on selected reproductive parameters in rat. J. Toxicol. Environ. Health, 9:677:687.

Lee, M.C., K. Voigt, L. Murray, and M. Lane. 1997. Environmental benefits and issues, engineering studies and risk analysis of mitigation options to reduce mine derived effects on the Ok Tedi/Fly River system, Papua New Guinea. Ok Tedi Mining Limited, Tabubil, Papua New Guinea. 25 pp.

Lehoczky, E., I. Szabados, R. Marth. 1996. Cadmium content of plants as affected by soil cadmium concentration. 27(5-8):1765-1777.

Le Bell, S., M. Salas, P. Chardonnet, and M. Bianchi. 1996. Rusa deer (*Cervus timorensis russa*) farming in New Caledonia: impact of different feed levels on herd breeding rate and performance of newborn fawns. Aust. Vet. J. 75:199:203.

Leino, T. and M. Lodenius. 1995. Human hair mercury levels in Tucurui area, State of Para, Brazil. The Science of the Total Environment. 175:119-125.

Lenka, M., K.K. Panda, and B.B. Panda. 1990. Studies on the ability of water hyacinth (*eichhornia crassipes*) to bioconcentrate and biomonitor aquatic mercury. Environ. Pollut. 0269-7491, 89-99.

Lepp, N. W. 1981a. Effect of heavy metal pollution on plants, Vol. 1. Effect of Trace Metals on Plant Function. Applied Science Publishers, London.

Lepp, N.W. 1981b. Copper. *In*: Effect of heavy metal pollution on plants. N.W. Lepp (ed). Volume 1. Chapter 4. Applied Science Publishers., NJ.

Lindsay, D.M. and J.G. Sanders. 1989. Arsenic uptake and transfer in a simplified estuarine food chain. Environmental Toxicology and Chemistry, Vol. 9, pp. 391-395.

Lindsay, W.L. 1979. Chemical equilibria in soils. New York: John Wiley and Sons.

Lodenius, M., A. Seppanen, and S. Autio. 1987. Sorption of mercury in soils with different humus content. Bull. Environ. Contam. Toxicol. 39:593-600.

Lodenius, M. and S. Autio. 1989. Effects of acidification on the mobilisation of cadmium and mercury from soils. Arch. Environ. Contam. Toxicol. 18:261-268.

Long, E.R. and L.G. Morgan. 1990. The potential for biological effects of sediment-sorbed contaminants tested in the National Status and Trends Program. NOAA Technical Memorandum NOS OMA 52. National Oceanic and Atmospheric Administration, Seattle, Washington.

Long, E.R., D.D. MacDonald, S.L. Smith, and F.D. Calder. 1995. Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. Env. Manag. 19:81-97.

Long, E.R., L.J. Field, and D.D. MacDonald. 1998. Predicting toxicity in marine sediment with numerical sediment quality guidelines. Environ. Toxicol. Chem. 17(4): 714-727.

Luwe, M.W.F. 1995. Distribution of nutrients and phytotoxic metal ions in the soil and in two forest floor plant species of a beech (*Fagus sylvatica* L.) stand. Plant and Soil. 168-169:195-202.

Mackay, R. D. 1976. New Guinea. Time-Life Books, Amsterdam.

Mackenzie, R.D., R.U. Byerrum, C.F. Decker, C.A. Hoppert, and R.F. Langham. 1958. Chronic toxicity studies, II. Hexavalent and trivalent chromium administered in drinking water to rats. Am. Med. Assoc. Arch. Ind. Health. 18:232-234.

Magurran, A. E. 1988. Ecological diversity and its measurement. Princeton University Press, Princeton, New Jersey.

Marsh, D.O., T.W. Clarkson, G.J. Myers, P.W. Davidson, C. Cox, E. Cernichiari, M.A. Tanner, W. Lednar, C. Shamlaye, O. Choisy, C. Hoareau, and M. Berlin. 1995. The Seychelles study of fetal methylmercury exposure and child development: Introduction. Neurotoxicology. 16(4):583-596.

Marshall, A. and M.T. Rau. 1999. Lower Ok Tedi and Middle Fly estimate of current vegetation dieback, classification of floodplain vegetation and prediction of maximum potential dieback extent. Part 1 of vegetation dieback extent, classification, and prediction: Lower Ok Tedi and Middle Fly, Western Province, Papua New Guinea. Environment Department, Ok Tedi Mining Ltd. Report. April and 4 July.

Mason, R.P., F.M.M. Morel, and H.F. Hemond. 1995a. The role of micro-organisms in elemental mercury formation in natural waters. Water, Air, and Soil Pollution. 80:775-787.

Mason, R.P., J. Hurley, A.K. Hanson, Jr., P.L. Donaghay, and J.M. Sieburth. 1993. Mercury biogeochemical cycling in a stratified estuary. Limnology and Oceanography. 38:1227-1241.

Mason, R.P., J.R. Reinfelder, and F.M.M. Morel. 1995b. Bioaccumulation of mercury and methylmercury. Water, Air, and Soil Pollution. 80:915-921.

Masscheleyn, P.H., R.D. Delaune, and W.H. Patrick. 1991. Effect of redox potential and pH on arsenic speciation and solubility in a contaminated soil. Environ Sci. Technol. 25: 1414-1419.

Matthews, A.D. 1983. Mercury content of commercially important fish of the Seychelles and hair mercury levels of a selected part of the population. Environmental Research. 30:305-312.

Matthews, C.K. and K.E. van Holde. 1990. Biochemistry. The Benjamins/Cummings Publishing Company, Redwood City, CA.

Matuk, Y., M. Ghosh, and C. McCulloch. 1981. Distribution of silver in the eyes and plasma proteins of the albino rat. Can J. Ophthalmol. 16:145-150.

McBride, M.B., L.D. Tyler, and D.A. Horde. 1981. Cadmium adsorption by soils and uptake by plants as affected by soil chemical properties. Soil Sci. Soc. Am. J. 45:739-744.

McGrath, S. P. 1995. Chromium and Nickel. *In*: Heavy metals in soils. B. J. Alloway ed. Blackie Academic & Professional Publishers.

Meador, J.P. 1991. The interaction of pH, dissolved organic carbon, and total copper in the determination of ionic copper and toxicity. Aquat. Toxicol. 19:13.

Mehring, A.L., J.H. Brumbaugh, A.J. Sutherland, and H.W. Titus. 1960. The tolerance of growing chickens for dietary copper. Poult. Sci. 39:713-719.

Mehrtens, J.M. 1987. Living snakes of the world. Sterling Publishing Co., New York.

Mo, S.C., D.S. Choi, and J.W. Robinson. 1988. A study of the uptake by duckweed of aluminium, copper, and lead from aqueous solution. J. Environ. Sci. Health, A23(2), 139-156.

Montague, J.J. 1984. Morphometric analysis of *Crocodylus novaeguineae* from the Fly River drainage, Papua New Guinea. Aust. Wildl. Res., 11:395:414.

Moreno, J.L., C. Garcia, T. Hernandez, and J.A. Pascual. 1996. Transference of heavy metals from a calcareous soil amended with sewage-sludge compost to barley plants. Bioresource Technology. 55:251-258.

Morren, G. E. B. 1977. From hunting to herding: Pigs and Control of Energy in Montane New Guinea. *In:* Bayliss-Smith, T. P. and R. G. Feacham eds. Subsistence and Survival: Rural Ecology in the Pacific. New York: Academic Press.

Morrison, D.W. 1980. Efficiency of food utilization by fruit bats. Oecologia, 45:270273.

MUDS. 1991. Multi-disciplinary technical study. Report to OTML. October 1991.

Muramoto, S., Y. Oki, H. Nishizaki, and I. Aoyama. 1989. Variation in some element contents of water hyacinth due to cadmium or nickel treatment with or without anionic surface active agents. J. Environ. Sci. Health, A24(8), 925-934.

Nakagawa, R. 1995. Concentration of mercury in hair of Japanese people. Chemosphere. 30(1): 127-133.

Nelson, H., D. Benoit, R. Erickson, V. Mattson, and J. Lindberg. 1986. The effects of variable hardness, pH, alkalinity, suspended clay, and humics on the chemical speciation and aquatic toxicity of copper. U.S. EPA, Environmental Research Laboratory, Duluth, Minnesota.

Nir, R., A. Gasith, and A.S. Perry. 1990. Cadmium uptake and toxicity to water hyacinth: effect of repeated exposures under controlled conditions. Bull. Environ. Contam. Toxicol. 44:149-157.

Norrish, K. 1975. The geochemistry and mineralogy of trace elements. *In*: Nicholas, D.J.D. and A. R. Egan eds. Trace Elements in Soil - Plant - Animal Systems. Academic Press. NY. pp. 55-82.

North, D.W. 1998. Risk assessment using the Taiwan data base: The need for further research. North Works, Inc. Unpublished Manuscript.

Norton, B. G. 1987. Why preserve natural variety? Princeton University Press, Princeton, New Jersey.

Nowak, R. M. 1994. Walker's bats of the world. The Johns Hopkins University Press, Baltimore.

NRC (National research Council). 1989. Recommended dietary allowances. 10th ed.Subcommitee on the Tenth edition of the RDAs, Food and Nutrition Board, Commission on Life Sciences, National Research Council. National Academy Press, Washington D.C.

NRCC (National Research Council of Canada). 1976. Effects of chromium in the Canadian environment. (NRCC No. 15017). Associate Committee on Scientific Criteria for Environmental Quality, Ottawa.

NRCC (National Research of Council Canada). 1979. Effects of mercury in the Canadian environment. Associate Committee on Scientific Criteria for Environmental Quality Subcommittee on Heavy Metals and Certain Other Elements. Publ. No. 16739.

Nriagu, J. 0. 1979. The global copper cycle. In: Copper in the Environment Part 1: Ecological Cycling, ed. J.O. Nriagu. Toronto: John Wiley and Sons.

ORNRL (Oak Ridge National Research Laboratory). 1995. Toxicological benchmarks for potential contaminants of concern for effects on soil and litter invertebrates and heterotrophic process. ES/ER/TM-126/R1. Environmental Restoration Risk Assessment Program, Oak Ridge, Tennessee

ORNRL (Oak Ridge National Research Laboratory). 1996. Toxicological benchmarks for wildlife: 1996 Revision. ES/ER/TM-86/R3. Health Sciences Research Division, Oak Ridge, Tennessee.

O'Connor, T.P., K.D. Daskalakis, J.L. Hyland, J.F. Paul, and J.K. Summers. 1998. Comparisons of sediment toxicity with predictions based on chemical guidelines. Environ. Toxicol. Chem. 17(3):468-471.

Odum, E. P. 1993. Ecology and our endangered life support systems, 2nd ed. Sinaue Associates, Inc., Sunderland, Massachusetts.

Ohlendorf, H. 1989. Bioaccumulation and effects of selenium in wildlife. Soil Science Society of America and American Society for Agronomy, Selenium in Agriculture and the Environment, SSSA Special Publication No. 23

Ondreicka, R., E. Ginter, and J. Kortus. 1966. Chronic toxicity of aluminium in rats and mice and its effects on phosphorus metabolism. Brit. J. Indust. Med. 23:305-313.

OTML (Ok Tedi Mining Limited). 1983. Environmental monitoring programme 6 monthly report January-June 1983. OTML Report PA/03/83-4, OTML, Tabubil.

OTML (Ok Tedi Mining Limited). 1984a. Environmental monitoring programme 6 monthly report June-December 1983. OTML Report PA/03/84-8, OTML, Tabubil.

OTML (Ok Tedi Mining Limited). 1984b. Environmental Monitoring Programme 6 Monthly Report January-June 1984. OTML Report PA/03/84-31, OTML, Tabubil.

OTML (Ok Tedi Mining Limited). 1985a. Environmental Monitoring Programme 6 Monthly Report July-December 1984. OTML Report PA/03/85-5, OTML, Tabubil.

OTML (Ok Tedi Mining Limited).. 1985b. Environmental Monitoring Programme 6 Monthly Report January-June 1985. OTML Report PA/03/85-14, OTML, Tabubil.

OTML (Ok Tedi Mining Limited). 1986. Environmental Monitoring Programme 6 Monthly Report July-December 1985. OTML Report ENV/86-04, OTML, Tabubil.

OTML (Ok Tedi Mining Limited). 1996. Environmental, financial, and risk analysis of various dredging and tailings storage schemes to mitigate mining impacts in the Ok Tedi/Fly River system. Ok Tedi Mining Limited Report No. ENV/96-08. December 1996.

OTML (Ok Tedi Mining Limited). 1997a. Hydrology annual report, 1995-1996. Prepared by A.J. Markham and A.S. Moi, Ok Tedi Mining Limited. Report No. ENV/97-02.

OTML (Ok Tedi Mining Limited). 1997b. Environmental chemistry section annual report - 1995-1996. Environment Department, Ok Tedi Mining Limited. Report No. ENV-97-03.

OTML (Ok Tedi Mining Limited). 1998a. Biology annual report, 1996-1997. Prepared by S. Swales, B. Figa, C. Tenakanai, K. Bakowa and A. Storey. Ok Tedi Mining Limited. Report No. ENV/98-04.

OTML (Ok Tedi Mining Limited). 1998b. APL compliance and additional environmental monitoring program annual report – 1996-1997. Environment Department, Ok Tedi Mining Limited. Report ENV/98-01

OTML (Ok Tedi Mining Limited). 1998c. Environmental chemistry section annual report – 1996-1997. Environment Department, Ok Tedi Mining Limited. Report No. ENV/98-02.

Otto, N.E. and P.F. Enger. 1960. Some effects of suspended sediment on growth of submersed pondweeds. Gen. Lab. Rep. No. Gen-27. U.S. Department of the Interior, Bureau of Reclamation, Denver, Colorado. 18pp.

Pagenkopf, G.K. 1983. Gill surface interaction model for trace-metal toxicity to fishes: role of complexation, pH and water hardness. Environ. Sci. Technol. 17: 347.

Paine, R. T. 1966. Food web complexity and species diversity. American Naturalist 100:65-75.

Panda, K.K., M. Lenka, and B.B. Panda. 1992. Monitoring and assessment of mercury pollution in the vicinity of a chloralkali plant. II. Plant-availability, tissue-concentration and genotoxicity of mercury from agricultural soil contaminated with solid waste assessed in barley (*Hordeum vulgare* L.). Environmental Pollution. 76:33-42.

Parametrix. 1997. Pinal Creek WQARF site: Preliminary remediation performance goals for aluminium, cobalt, and manganese. Parametrix, Kirkland, Washington. 18pp. + appendices.

Parkhurst, B.R., W. Warren-Hicks, R.D. Cardwell, J. Volosin, T. Etchison, J.B. Butcher, and S.M. Covington. 1996. Methodology for aquatic ecological risk assessment: A multi-tiered approach. Project 91-AER-1. Prepared for Water Environment Research Foundation, Alexandria, Virginia.

Part, P. and O. Svanberg. 1981. Uptake of cadmium in perfused rainbow trout (*Salmo gairdneri*) gills. Can. J. Fish. Aquat. Sci. 38: 917-924.

Pascoe, G.A., R.J. Blanchet, and G. Linder. 1994. Bioavailability of metals and arsenic to small mammals at a mining waste-contaminated wetland. Archives of Environmental Contamination and Toxicology, 27:44-50.

Petersen, P.J. and C.A. Girling. 1981. Other Trace Metals. *In:* Lepp, N.W. (ed). Effect of heavy metal pollution on plants. Volume 1. Chapter 7. Applied Science Publishers., NJ.

Playle, R.C. and D.G. Dixon. 1993. Copper and cadmium binding to fish gills: modification by dissolved organic carbon and synthetic ligands. Can. J. Fish. Aquat. Sci. 50: 2667-2677.

PNG. 1998. Recommended water quality standards for Papua New Guinea. Revised Draft – (2). Department of Environment and Conservation, Boroko, Papua New Guinea.

Prendergast, J.B., M.T. Rau, A.S. Moi, and A. Marshall. 1996. An Evaluation of the effectiveness of mitigation works in reducing dieback on the Ok Tedi and Fly River floodplains. Environmental, Financial, and Risk Analysis of Various Dredging and Tailings Storage Schemes to Mitigate Mining Impacts in the Ok Tedi/Fly River System. Ok Tedi Mining Limited. Company Report No. ENV96-08.

Presant, E. W. and Tupper, W. M. 1963. Can. J. Soil. Sci. 45: 305-310. (In: Alloway 1995, Ch. 14.4).

PTI (Envoronmental Services). 1996. Interim results from a study of the chemical composition, limnology, and ecology of three existing Nevada pit lakes. Letter to P. Maley, May 17, 1996.

R&D Environmental Pty Ltd. 1998. Review of the biology of the Ok Tedi/Fly River system. R&D Environmental, Australia. 73 pp.

Rabitsch, W.B. 1995. Metal accumulation in arthropods near a lead/zinc smelter in Arnoldstein, Austria. I. Environmental Pollution. 90(2):221-237.

Rai, D., L.E. Eary, and J.M. Zachara. 1989. Environmental chemistry of chromium. Sci. Total Environ. 86:15-23.

Rand, A.L. and E.T. Gilliard. 1968. Handbook of New Guinea Birds. Natural History Press, New York

Rand, G.M. and S.R. Petrocelli. 1985. Fundamentals of aquatic toxicology. Hemisphere Publishing Corp., New York, New York. 666pp.

Rand, G.M. 1995. Fundamentals of aquatic toxicology: Effects, environmental fate and risk assessment. Second Edition. Taylor and Francis, Washington D.C. 1125pp.

Rau, M. T. 1994. Overbank flooding effects on vegetation in the lower Ok Tedi. Ok Tedi Mining Limited.

Rau, M. T. 1995. Overbank flooding effects on vegetation in the lower Ok Tedi. Ok Tedi Mining Limited, Port Moresby, Papua New Guinea.

Rau, Monica. 1998. Personal communication to Doug Reagan.

Reagan, D. P. In Press. Determining ecological values: a critical step in assessing ecological risk, Chapter 6. *In:* D. Paustenbach, ed. Health risk assessment. John Wiley and Sons, Inc., New York.

Reagan, D.P. and R. B. Waide. 1996. The food web of a tropical rain forest. University of Chicago Press, Chicago.

Reagan, D.P. and X. Silva del Poso. 1995. Using biodiversity methods to assess the impacts of oil and gas development in tropical rain forests. Bulletin of the Ecological Society of America 76:222

Reagan, D. P., G.R Camilo, and R.B. Waide. 1996. The community food web: Major properti.es and patterns of organization. *In:* Reagan D.P. and R.B. Waide, eds. The Food Web Of A Tropical Rain Forest, Chicago University Press, Chicago, Chapter 14 pp 461-510.

Reid, B. 1987. Food intake and growth rate of cassowary chicks, *Casuarius* sp., reared at Mendi, Southern Highland Papua New Guinea. Int. Zool. Yb., 26:189-198.

Reiger, H. A. 1993. The notion of natural and cultural integrity. *In:* Woolly S., J. Kay, and G. Francis, eds. Ecological integrity and the Management of Ecosystems. St. Lucia Press, Ottawa.

Romney, E. M., A.Wallace, R.Wood, A.M. El-Gazzar, J.V. Childress, and J.V. Alexander. 1979. Commun. Soil. Sci. Plant Anal. 8: 719-725. *In:* Alloway 1995, Ch 14.4.

RTECS (Registry of Toxic Effects of Chemical Substances). 1997. On-line computer database. NIOSH: U.S. Dept of Health and Human Services, Center for Disease Control.

Rudd, J.W.M., M.A. Turner, A. Fuutani, A.L. Swick, and B.E. Townsend. 1983. The English-Wabigoon river system: A synthesis of recent research with a view towards mercury amelioration. Can. J. Fish. Aq. Sci. 40:2206-2217.

Sachdev, P., W.L. Lindsay, and D.L. Deb. 1992. Activity measurements of zinc in soils of different pH using EDTA. Geoderma 55:247-257.

Sadiq, M. 1992. Toxic metal chemistry to marine environments. Marcel Dekker, Inc., New York, New York.

Sadiq, M. and Enfield, C. G. 1984a. Soil Science. 138: 262-270. (In: Alloway 1995, Ch. 7).

Sadiq, M. and Enfield, C. G. 1984b. Soil Science. 138: 335-340. (In: Alloway 1995, Ch. 7).

Saeki, K., M.Fujimoto, D. Kolinjim, and R. Tasukawa. 1996. Mercury concentrations in hair from populations in Wau-Bulolo area, Papua New Guinea. Archives of Environmental Contamination and Toxicology. 30: 412-417.

Schlicker, S.A. and D.H. Cox. 1968. Maternal dietary zinc, and development and zinc, iron, and copper content of the rat fetus. J. Nutr. 95:287-294.

Schoof, R.A., L.A. Yost, E. Crecelius, K. Irgolic, W. Goessler, H.R. Guo, and H. Greene. 1998. Dietary arsenic intake in Taiwanese districts with elevated arsenic in drinking water. Human and Ecological Risk Assessment. 4(1); 117-135.

Schoof, R.A., L.A.Yost, J.Eickhoff, E.A. Crecelius, D.W. Cragin, D.M. Meacher and D.B. Menzel. 1999. A market basket survey of inorganic arsenic in food. *In*: Press Food and Chemical Toxicology, accepted February 1999.

Schroeder, H.A., and M. Mitchener. 1971. Toxic effects of trace elements on the reproduction of mice and rats. Arch Environ Health 23:102-106.

Sekhran, N. and S. Miller, eds. 1995. Papua New Guinea Country Study on Biological Diversity. Africa Centre for Resources and Environment.

Sela, M., J. Garty, and E. Tel-or. 1988. The accumulation and the effect of heavy metals on the water fern *Azolla filiculoides*. New Phyto. (1989), 112, 7-12.

Shaw, T.L. and V.M. Brown. 1974. The toxicity of some forms of copper to rainbow trout. Water Resources. 8:377-382.

Sheppard, S.C., W.G. Evenden, S.A Abboud, and M.Stephensen. 1993. A plan life-cycle bioassay for contaminated soil, with comparison to other bioassays: Mercury and Zinc. Arch. Environ. Contam. Toxicol. 25: 27-35.

Shuman, L.M. 1975. The effect of soil properties on zinc adsorption by soils. Soil Sci. Soc. Am. Proc. 39:454-459.

Sinha, S., M. Gupta, and P. Chandra. 1994. Bioaccumulation and toxicity of Cu and Cd in *Vallisneria spiralis* (L.). Environ. Monitor. Assess. 33:75-84.

Slayton, T.M., B. Beck, K.A. Reynolds, S.D. Chapnick, L.A. Yost, R.A. Schoof, T.D. Gauthier, and L. Jones. 1996. Environmental Health Perspective-correspondence. 104(10):1-12-1013.

Slooff, W., B.J.A. Haring, J.M. Hesse, J.A. Janus, and R. Thomas, eds. 1990. Integrated criteria document: Arsenic. Report No. 710401004. Research for Man and Environment (RIVM), Bilthoven, The Netherlands.

Slooff, W., R.F.M.J. Clevan, J.A. Janus, and J.P.M. Ros. 1989. Integrated criteria document copper. Report No. 758474009. National Institute of Public Health and Environmental Protection. Bilthoven, Netherlands.

Snowden, R.E.D. and B.D. Wheeler, 1995. Chemical changes in selected wetland plant species with increasing Fe supply, with specific reference to root precipitates and Fe tolerance. New Phytol. 131: 503-520.

Sorentino, C. 1979. Mercury in marine and freshwater fish of Papua New Guinea. Australian Journal of Marine Freshwater Research. 30:617-623.

Sparling, D.W. and T.P. Lowe. 1996. Environmental hazards of aluminum to plants, Invertebrates, Fish and Wildlife. Rev. of Environ. Contam. and Toxicol. 145: 1-127.

Spry, D.J., P.V. Hodson, and C.M. Wood. 1988. Relative contributions of dietary and waterborne zinc in the rainbow trout, *Salmo gairdneri*. Can. J. Fish. Aquat. Sci. 45:32-41.

Stahl, J.L., J.L. Greger, and M.E. Cook. 1990. Breeding-hen and progeny performance when hens are fed excessive dietary zinc. Poult. Sci. 69:259-263.

Stara, J.F. and L.S. Erdreich. 1984. Selected approaches to risk assessment for multiple chemical exposures: progress report on guideline development at ECAO (Environmental Criteria and Assessment Office)-Cincinnati. EPA, Environmental Criteria and Assessment Office, Cincinnati, Ohio. EPA Report No. EPA/600/9-84/014A, NTIS No. PB84-226992.

Stauber, J.L. and S.C. Apte. 1996. Bioavailability of copper to algae in the Fly River system, PNG. CSIRO Investigation Report CET/IR 464 R. Prepared for Ok Tedi Mining Limited. 78 pp.

Steinnes, E. 1995. Mercury. *In*: Alloway B.J., ed. Heavy metals in soils, 2nd edition. New York: John Wiley and Sons, Inc.

Stephan, C.E., D.I. Mount, D.J. Hansen, J.H. Gentile, G.A. Chapman, and W.A. Brungs. 1985. Guidelines for deriving numerical national water quality criteria for the protection of aquatic organisms and their uses. U.S. Environmental Protection Agency, Washington, D.C. NTIS No. PB85-227049. 98 pp.

Storey, A.W. 1998. Review of dietary data for fish from the Fly River system: A precursor to constructing a food web. Prepared for OK Tedi Mining Ltd.

Storey, A.W. and R.E.W. Smith. 1998. Food webs for the Fly River fish fauna. Prepared for Ok Tedi Mining Limited. 19 pp.

Sullivan, M.F., J.T. Hardy, B.M. Miller, R.L. Buschbom, and T.C. Siewick. 1984. Absorption and distribution of cadmium in mice fed diets containing either inorganic or oyster-incorporated cadmium. Toxicol. App. Pharmacol. 72:210-217.

Sunda, W.G. and R.R.L. Guillard. 1976. The relationship between cupric ion activity and the toxicity of copper to phytoplankton. J. Mar. Res. 34: 511.

Suter II, G.W. 1990. Endpoints for regional ecological risk assessments. Environmental Management. 14(1):19-23.

Suter II, G.W., M.A. Futrell, and G.A. Kerchner. 1992. Toxicological benchmarks for screening of potential contamination of concern for effects on aquatic biota on the Oak Ridge Reservation, Oak Ridge, Tennessee. Oak Ridge National Laboratory, Oak Ridge, Tennessee. ORNL/ER—139, DE93 000719. 42 pp. + appendices.

Sutton, S. L. 1989. The spatial distribution of flying insects. Chapter 24 In H. Lieth and M.J.A. Werger eds. Ecosystems of the World 14b: Tropical Rain Forest Ecosystems. Elsevier Scientific Publications, Amsterdam.

Swales, S. 1997. Theoretical yields and current status of commercial and artisanal fisheries in the middle Fly River. Environmental Department, Ok Tedi Mining Limited.

Taylor, J.A. 1979. The foods and feeding habits of subadult *Crocodylus porosus* Schneider in Northern Australia. Australian Wildlife Research, 6:347-359.

TERA (Toxicology Excellence for Risk Assessment). 1998. International toxicity estimates for risk: A World Wide Web toxicity database Sponsored by: Toxicology Excellence for Risk Assessment (TERA). Available at: http://www.tera.org.

TERA (Toxicology Excellence for Risk Assessment). 1999. International toxicity estimates for risk: Non-Cancer Oral Toxicity of Inorganic Arsenic. A World Wide Web toxicity database Sponsored by: Toxicology Excellence for Risk Assessment (TERA). Available at: http://www.tera.org

Terborgh, J. 1992. Maintenance of diversity in tropical forests. Biotropica 24:283-292.

Terborgh, J. and C. P. van Schaik. 1997. Minimizing species loss: the imperative of protection. In R. Kramer, C. van Scaik, and J. Johnson eds. Last Stand: Protected Areas and the Defense of Tropical Biodiversity. Oxford University Press, Oxford.

Thomas, D.G., L.G. Solbe, J. Kay, and A. Cryer. 1983. Environmental cadmium is not sequestered by metallothionine in rainbow trout. Biochemical and Biophysical Research Communications. 110:594-592.

Thomas, D.W. 1984. Fruit intake and energy budgets of frugivorous bats. Physiol. Zool., 57:457-467.

Thompson, S.E., C.A. Burton, D.J. Quinn, and Y.C. Ng. 1972. Concentration factors of chemical elements in edible aquatic organisms. University of California, Lawrence Livermore Laboratory, Biomedical Division. Livermore, California. UCPL-50564 Rev. 1. 73 pp.

Tinkerame, J.T. 1998. Personal communication of October 20, 1998 with Mr. Paul Seidel of Parametrix. Comments on moisture content of fish flesh samples collected in 1996.

Townsend, P. K. 1974. Sago Production in a New Guinea Economy, Hum. Ecol. 2: 217-236.

Tseng, W.P. 1977. Effects and dose-response relationships of skin cancer and blackfoot disease with arsenic. Environmental Health Perspectives 19: 109-119.

Tseng, W.P., H.M. Chu, S.W. How, J.M. Fong, C.S. Lin, and S. Yeh. 1968. Prevalence of skin cancer in an endemic area of chronic arsenicism in Taiwan. Journal of National Cancer Institute. 40: 453-463.

U. S. Fish and Wildlife Service. 1964. Pesticide-wildlife studies, 1963: a review of Fish and Wildlife Service investigations during the calendar year. FWS Circular 199.

UNEP (United Nations Environment Programme). 1995. Environmental impact of large-scale mining in Papua New Guinea: Mining residue disposal by the Ok Tedi Copper-Gold Mine. Selbstverlag Fachbereich Geowissenschaften, Berlin, Germany.

van den Berg, G.A., J.P. Gustav Loch, L.M. van der Heijdt, and J.J.G. Zwolsman. 1998. Vertical distribution of acid-volatile sulfide and simultaneously extracted metals in a recent sedimentation area of the River Meuse in The Netherlands. Environ. Toxicol. Chem. 17(4):758-763.

Vymazal, J. 1990. Uptake of lead, chromium, cadmium, and cobalt by *Cladophora glomerata*. Bull. Environ. Contam. Toxicol. 44:468-472.

Weber, J.H. 1993. Review of possible paths for abiotic methylation of mercury in the aquatic environment. Chemosphere. 26:2063-2077.

Weber, M.B., H. Schat, and W.M. Ten Bookum-Van Der Maarel. 1991. The effect of copper toxicity on the contents of nitrogen compounds in Silene vulgaris (Moench) Garcke. Plant and Soil. 133: 101-109.

Weres, O., A.R. Jaouni, and L. Tsao. 1989. The distribution, speciation and geochemical cycling of selenium in a sedimentary environment, Kesterson Reservoir, California, USA. Appl. Geochem. 4: 543-563.

White, D.H. and M.T. Finley. 1978. Uptake and retention of dietary cadmium in mallard ducks. Environ. Res. 17:53-59.

WHO (World Health Organization). 1972. World Health Organization technical Report Series, No. 505. Evaluation of certain food additives and the contaminants: mercury, lead and cadmium. Sixteenth Report of the Joint FAO/WHO Expert Committee of Food Additives, Geneva, April 4-12 1972.

WHO (World Health Organization). 1986. Environmental health criteria 59; principles for evaluating health risks from chemicals during infancy and early childhood: The need for a special approach. IPCS: International Programme on Chemical Safety. Published under the Joint sponsorship of the United Nations Environmental Programme, the International Labour Organization, the World Health Organization, and on behalf of the Commission of the European Communities, World Health Organization, Geneva.

WHO (World Health Organization). 1991. Inorganic mercury. Environmental Health Criteria 118. Geneva, Switzerland.

WHO (World Health Organization). 1996a. Guidelines for drinking water quality. Volume 2 – Health criteria and other supporting information. Second Edition. ICPS, World Health Organization.

WHO (World Health Organization). 1996b. Trace elements in human nutrition and health. Prepared in collaboration with food and agriculture organization of the United Nations and the International Atomic Energy Agency. World Health Organization, Geneva.

Wilson, J.D. 1996. Thresholds for carcinogens: A Review of the relevant science and its implications for regulatory policy. Discussion Paper 96-21. Published By Resources for the Future, Washington D.C. Available at: http://www.rff.org/methods/risk.htm.

Winner, R.W. and J.D. Gauss. 1986. Relationships between chronic toxicity and bioaccumulation of copper, cadmium and zinc as affected by water hardness and humic acid. Aquat. Toxicol. 8: 149-161.

Woodward, D.F., J.A. Hansen, H.L. Bergman, E.E. Little, and A.J. DeLonay. 1995. Brown trout avoidance of metals in water characteristic of the Clark Fork River, Montana. Can. J. Fish. Aq. Sci. 52:2031-2037.

Woodward, K.B. and A. Dunning. 1992. Production cycles and characteristics of rusa deer in Queensland, Australia. In: The Biology of Deer, Robert D. Brown Ed. Springer-Verlag Press, New York, New York.

Woodyard, D.E. and J.B. Haufler. 1991. Evaluation for sludge-borne elements to Wildlife Food chains. Garland Publishing, Inc., New York and London.

Woolson, E.A. 1973. Arsenic phytotoxicity and uptake in six vegetable crops. Weed Sci. 21: 524-527.

Woolson, E.A. 1975. Arsenical Pesticides. American chemical society symposium. Vol. 7. American Chemical Society, Washington. 176 pp.

Wu, L., J. Chen, K.K. Tanji, and G.S. Banuelos. 1995. Distribution and biomagnification of selenium in a restored upland grassland contaminated by selenium from agricultural drain water. Environ. Toxicol. Chem. 14(4):733-742.

Yaru, B.T. 1998. Is mine-derived sediments exacerbating hypoxia in off-river water bodies of the Fly River system? Draft version. Niugini Environmental Consultants, Ltd., Artarmon, Australia. 18 pp.

Yok, D. 1989. Demographic and dietary investigation along the Strickland, Lagaip and Porgera Rivers. PLACER (P.N.G.) Pty. Limited. Hydrology and Environmental Section.

Yok, D. 1990. Subsistence fisheries and demographic investigations within the Lake Murray area: Buseki, Usukof, Miwa and Kusikina villages. October/November 1989. Placer (P.N.G.) Pty Limited., PJV.

Yok, D. 1990a. Demographic and dietary investigation along the Strickland River: Tiumsiniwam Village. Placer (P.N.G.) Pty Ltd., PJV. 16 pp.

Yok, D. 1990b. Subsistence fisheries and demographic investigations within the Lake Murray area: Buseki, Usukof, Miwa and Kusikina villages. October/November 1989. Placer (P.N.G.) Pty Limited., PJV.

Yost, L.A., R.A. Schoof, and R. Aucoin. 1998. Intake of inorganic arsenic in the North American diet. Hum. Ecol. Risk Assess. 4(1):137-152.

Zachara, J.M., C.C. Ainsworth, C.E. Cowan, and C.T. Resch. 1989. Adsorption of chromate by subsurface soil horizons. Soil Sci. Soc. Am. J. 53:418-428.

Zamuda, C.D. and W.G. Sunda. 1982. Bioavailability of dissolved Cu to the American oyster Crassostrea virginica. I. Importance of chemical speciation. Mar. Biol. 66: 77-82.

Zaranyika, M.F., F. Mutoko, and H. Murahwa. 1993. Uptake of Zn, Co, Fe, and Cr by water hyacinth (*Eichhornia crassipes*) in Lake Chivero, Zimbabwe. The Science of the Total Environment 153 (1994), 117-121.

Zweifel, R. G. 1972. A revision of the frogs of the subfamily Asterophryinae, family Microhylidae. Bulletin of the America Museum of Natural History, Volume 148, Article 3.

8. GLOSSARY

Acid Rock Drainage (ARD)	Acid rock drainage refers to low pH groundwater or leachate with high metal levels that drain from mine waste rock and tailings.
Acute Exposure	One dose or multiple doses occurring within a short time (1 –7 days).
Acute Toxicity	Significant probability of mortality or other effects from short-term (often 96 hr), relatively high concentration exposure to toxic chemicals (Rand 1995).
Acute-Chronic Ratio (ACR)	The ratio of a chemical's acute toxicity to its chronic toxicity for the same species (Rand 1995).
Additive effects	The potential for adverse effects on health due to the combined action of two or more chemicals which have a similar mode of action. It assumes that the combined effect of the subthreshold effects of several chemicals could result in an adverse effect.
Assessment Endpoint	Explicit expressions of the actual environmental or societal value to be protected, or the undesired effect whose probability of occurrence is estimated in a risk assessment. Examples include extinction of an endangered species, eutrophication of a lake, or the damage to a fishery by water pollution (Parkhurst et al. 1995).
Averaging Time	Period over which exposure is averaged (years or days). For non-carcinogens exposure is typically averaged over the duration of exposure. For carcinogens exposure is averaged over an estimate of lifespan.
Background	Chemical concentrations or intakes originating from chemical concentrations in local environmental media unimpacted by mine tailings, leachate, or other activities. Background concentrations are used to characterize ambient conditions unrelated to the mine.
Bioaccumulation	The amount of chemical taken up by the organism attributable to both bioconcentration and dietary accumulation (Rand 1995).
Bioavailability	The degree to which a chemical is available to the target organism or tissue.
Bioconcentration	The process whereby chemicals enter aquatic organisms through the gills or epithelial tissue directly from the water (Rand 1995).

Biomagnification	The process by which the tissue concentration of a bioaccumulated chemical increases as it passes up the food chain through at least two trophic levels (minimum of three involved) (Rand 1995).
Biota	Fish, aquatic invertebrates (e.g., prawns) and plants.
Carcinogenesis	The origin or production of cancer. The carcinogenic event so modifies the genome and/or other molecular control mechanisms in the target cells that these result in unregulated cell proliferation.
Carcinogenic (Cancer) Risk	Carcinogenic risk as used in this risk assessment refers to the increased incremental (over background) probability of cancer occurring in an exposed person or exposed population.
Carcinogenic	Having the ability to cause or promote cancer.
Characterization of Ecological Effects	The process for quantitatively defining the adverse effects on individuals, populations, and communities elicited from exposure (Parkhurst et al. 1995).
Characterization of Exposure	The process for quantitatively defining the expected environmental concentrations/doses (EECs and EEDs) and pathways to which the receptors are exposed (Parkhurst et al. 1995).
Chronic daily intake (CDI)	An estimate of the daily amount of chemical on a mass per unit body weight (i.e., mg/kg-day) taken into the body and available for absorption at an exchange boundary (e.g., lungs, gastrointestinal tract). The CDI is calculated as a daily average over the period of exposure and is used only to evaluate chronic (long-term) exposures.
Chronic Exposure	Multiple exposures occurring over an extended period of time, or a significant fraction of the animal's or the individual's lifetime up to the entire duration of life.
Chronic Toxicity	Significant probability of effects on growth, yield, reproduction or survival from long-term exposure to toxic chemicals (Rand 1995).
Community	An assemblage of populations of plants, animals, bacteria, and fungi that live in an environment and interact with one another, forming a distinctive living system with its own composition, structure, environmental relations, development, and function.
Conceptual Model	A written description and visual representation of predicted relationships between ecological entities and the stressors to which they may be exposed.
Cumulative	The total exposure or risk accruing from all sources (site-related and background).

Dietary Accumulation	The amount of chemical taken up by the organism that is solely attributable to dietary intake.
Dose	The mass of a substance given to an organism and in contact with an exchange boundary (e.g., gastrointestinal tract) per unit body weight per until time (e.g., mg/kg-day).
EC50	Median effect concentration, the concentration causing an effect (such as death, immobilization or serious incapacitation, decreased growth, and reproductive impairment) to fifty percent of the test organisms (Rand 1995).
Ecosystem	The biotic community and abiotic that interact within a specified location in space and time.
Excess Lifetime Risk	The additional or extra risk incurred over the lifetime of an individual by exposure to a toxic substance.
Expected Environmental Concentration	The estimated concentration of chemicals in surface water, sediments, and the food of fish and wildlife (Parkhurst et al. 1995).
Expected Environmental Dose	The estimated dose that a receptor will assimilate from consumption of water, vegetation, or fish and wildlife prey.
Exposure	The contact or co-occurrence of a stressor with a receptor.
Exposure Assessment	The determination or estimation (qualitative or quantitative) of the magnitude, frequency, duration, and route of exposure to chemicals in environmental media.
Exposure Frequency and Duration (EF or ED)	Estimates of how long and how often exposure occurs. EF in units of days/years; ED in units of years.
Exposure Pathway	The course a chemical or physical agent takes from a source to an exposed organism. An exposure pathway describes a unique mechanism by which an individual or population is exposed to chemicals or physical agents at or originating from a site. Each exposure pathway includes a source, or release from a source, an exposure point, and an exposure route. If the exposure point differs from the source, a transport/exposure medium (e.g., air) or media (in cases of intermedia transfer) is/are also included.
Exposure Route	The mechanism by which a chemical or physical agent comes in contact with an organism (i.e., by ingestion, inhalation, dermal contact).
Frugivore	A fruit-eating bird or mammal.

Gramnivore	A grain-eating bird or mammal.
Hazard Quotient (HQ)	The ratio of the concentration or dose of a chemical over the concentration or dose at which no adverse effects of any kind are expected. When HQs are less than one, negligible risks are expected.
LC50	Median lethal concentration; the concentration causing death to fifty percent of the test organisms (Rand 1995).
LD50	Median lethal dose; the dose causing death to fifty percent of the test organisms (Rand 1995).
Lifestage	Differing periods of life in which people have been categorized for analytical convenience. For the purposes of the Ok Tedi Human Health Risk assessments, lifestages are children (1-7 years old), adolescents (8-17 years old), and adults (older than 18 years).
Lifetime Average Daily Intake (LADI)	An estimate of the daily amount of chemical on a mass per unit body weight (i.e., mg/kg-day) taken into the body and available for absorption at an exchange boundary (e.g., lungs, gastrointestinal tract). The LADI is calculated as a daily average over an estimate of the lifespan and is used only to evaluate exposures to chemical carcinogens.
LOEC	Lowest observed effect concentration; the lowest concentration causing an effect (Rand 1995).
Lowest Observed Adverse Effect Level (LOAEL)	In dose-response experiments, the lowest exposure level at which there are statistically or biologically significant increases in frequency or severity of adverse effects between the exposed population and its appropriate control group.
Measurement Endpoint	An expression of an observed or measured response to a hazard; it is a measurable environmental characteristic that is related to the valued characteristic chosen as the assessment endpoint (Parkhurst et al. 1995).
Metalloenzymes	Enzymes which contain metals as an essential component (i.e., cofactor) of their structure.
Naturally Occurring Background Levels	Ambient concentrations of chemicals (e.g., aluminum, manganese) that are present in the environment and have not been influenced by humans.
Nectarivore	An organism that feeds on nectar and pollen.

No Observed Adverse Effect Level (NOAEL)	In dose-response experiments, an exposure level at which there are no statistically or biologically significant increases in frequency or severity of adverse effects between the exposed population and its appropriate control; some effects may be produced at this level, but they are not considered to be adverse, nor precursors to specific adverse effects. In an experiment with more than one NOAEL, the regulatory focus is primarily on the highest one, leading to the common usage of the term NOAEL to mean the highest exposure level without adverse effect.
NOEC	No observed effect concentration; the concentration causing no effect (Rand 1995).
Off-River Water Bodies	Tie channels, oxbow lakes, blocked valley lakes, swamps, lagoons, periodically flooded forest and any other body of water that is directly affected by the river but not part of the main stream.
Omnivore	An animal that feeds on both plants and animals.
Planktivore	An organism that feeds on plankton.
Population	A potentially interbreeding group of individuals of a single species.
Problem Formulation	The step where goals of the risk assessment are defined and exposure routes for stressors (chemicals) are identified.
Reference Dose (RfD) Chronic	An estimate (with uncertainty spanning perhaps an order of magnitude or greater) of a daily exposure level for the human population, including sensitive subpopulations, that is likely to be without an appreciable risk of deleterious effects during a lifetime. Chronic RfDs are specifically developed to be protective for long-term exposure to a compound (as a Superfund guideline, seven years to lifetime).
Risk	The likelihood of a prescribed undesired effect, such as injury, disease or death, resulting from human actions or a natural catastrophe (Parkhurst et al. 1995).
Risk Characterization	The process that defines the potential for or probability of adverse effects to the receptor population given exposure to a range of expected environmental concentrations (EECs).
Screening	A risk assessment process in which conservative estimates of exposure and toxicity are used to identify chemicals that pose negligible risks. Chemicals so identified are referred to as being “screened out”.

Slope factor	A slope factor is an estimate of the carcinogenic potency of a chemical developed by U.S. EPA under the assumption that there is no threshold (i.e., safe dose) for the carcinogen. Use of slope factors, together with LADIs, generate estimates of the probability of a carcinogenic response.
Stressor	Any physical, chemical, or biological entity that can induce an adverse response.
Stressors of Potential Concern (SOPCs)	Chemicals that have been identified by the balance of available evidence as posing potential risks to health and may have a relationship to mine tailings or processes.
Subchronic Exposure	Either multiple or continuous exposures occurring over a period of months but less than many years (chronic).
Tailings	The residual material remaining after ore has been milled for removal of metals, it may be considered finely ground rock.
Tier	A risk-assessment approach in which an initial screening is followed by additional data collection and analysis. Each level of re-analysis or follow-on analysis is a new Tier.
Trophic Levels	A functional classification of taxa within a community that is based on feeding relationships.
Uncertainty Factor (UF)	One of several, generally 10-fold, factors used in operationally deriving the toxicity threshold from experimental data. UFs are intended to account for (1) the variation in sensitivity among the members of the human population; (2) the uncertainty in extrapolating animal data to the case of humans; (3) the uncertainty in extrapolating from data obtained in a study that is of less-than-lifetime exposure; and (4) the uncertainty in using LOAEL data rather than NOAEL data.
Upper Bound	An estimate of the plausible upper limit to the true value of the quantity. This is usually not a statistical confidence limit.