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**Spatio-temporal dynamics and population biology of the  
Fly River Herring (*Nematalosa papuensis*): implications  
for freshwater lake management in Papua New Guinea.**

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Graduate Certificate in Research Methods (JCU)**

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In the School of Marine and Tropical Biology  
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## Abstract

In the face of continuous threats to the freshwater systems of the world from waste of anthropogenic origins and climate-induced environmental changes, the productivity of large floodplain ecosystems in virtually every continent is under serious threat of survival. Fish distributions and temporal dynamics are in part functions of habitat structure and conditions. Riverine fish population biology and dynamics have been studied extensively worldwide and described under various river productivity models that explain community dynamics and structure according to a range of spatial and temporal factors. Fish distribution and movements have been described in four dimensions – longitudinal, lateral, vertical, and temporal (seasonal) – that reflect the dynamic spatial and temporal nature of fish movements and habitat requirements in freshwater systems.

Much of what is known about the fish fauna of large tropical floodplain systems of the world is the result of work undertaken in the Amazonian, Asian and African wetlands. Very little information, however, is available on floodplain fisheries ecology of the Indonesian archipelago, the island of New Guinea or the South Pacific. This has led to this study being undertaken to understand the floodplain ecology of the Fly System in Papua New Guinea (PNG) by studying a key clupeoid species.

Though cold-water species dominate clupeoid fisheries globally, the greatest diversity of clupeids is found in the warmer marine and freshwaters of the tropics. The greatest diversity is found in the Indo-West Pacific region, although this constitutes a very small percentage of the world clupeoid catch, probably because their commercial potential have not have been fully explored, or because, in most cases, the focus of warm water fisheries is more on food security than cash economy.

The Fly River herring (*Nematalosa papuensis*), a clupeid of the Indo-Pacific region, is from the genera *Nematalosa* (subfamily Dorosomatinae) of freshwater systems in Papua New Guinea (PNG) and Australia. This species, a primary consumer, is a key species in the fisheries ecology of the heavily polluted Fly River system, making up to 38% of the fish biomass of the Fly River off-river water bodies (lakes), yet its natural history in the largest wetland system in Australasia and Pacific is largely unknown. This study investigated the habitat of *N. papuensis*, its population dynamics and structure, its diet, and its growth and reproduction biology in seven off river water bodies (ORWB) representing different habitats along the Fly River system. The habitats studies included 3 forested oxbows, a mixed forest and grassed fringed oxbow, a grassed floodplain oxbow, and 2 grassed floodplain blocked valley lakes. Six sampling trips were undertaken, 3 in 2008 (2 wet and 1 dry season) and 3 in 2009 (2dry and 1 wet season). Three replicate sites within each ORWB were sampled for fish, water quality, and aquatic macrophytes during each trip. No significant seasonal variations ( $p>0.05$ ) were detected in most of the parameters that were studied; however, there were significant habitat-influenced differences in the biology of this species, most probably in relation to impacts from

the Ok Tedi copper mine, upstream.

Water quality analyses revealed significant spatial variations at all sites. Water quality at lakes nearest to the Ok Tedi mine were more degraded with higher dissolved metal concentrations and total suspended solids in the water column. Significant habitat differences were observed in pH, temperature, dissolved oxygen, conductivity, turbidity and total suspended solids. There were no seasonal differences in pH, conductivity, and TSS; but significant seasonal differences in temperature, dissolved oxygen and turbidity were detected. Dissolved metal levels were generally low; with the exception of copper (Cu) where mean Cu concentrations exceeded guidelines (of 1.4 µg/L) at mine-impacted sites, being particularly high at sites closest to the mine. Forty-five aquatic macrophyte species were recorded. Grassed floodplain lakes had greater vegetation density and species richness than forested oxbows, and density and richness of macrophytes increased with distance from the mine.

During the study 17,361 adult and 2,164 juvenile *N. papuensis* were collected, with a combined total weight of 1008.1 kg. Fish biomass was significantly lower in forested oxbows than in grassed floodplain lakes. Most lakes showed a trend of higher catches during the dry than wet season; however, these seasonal variations were only significant for two grassed floodplain lakes. There were significant spatial but not seasonal variations in juvenile catches. Changes in habitat quality were reflected in the biology of *N. papuensis* populations, as population biomass increased with distance from the mine, indicating an effect of mine pollution. A lagged phase analysis of the long-term data against the *El Niño* Southern Oscillation Index (ENSO) showed that short-term climatic events such as *El Niños* were not significant contributors to the long-term declining trend in biomass.

Length-weight relationships (LWR) varied significantly between lakes and sexes. Separate analysis for each sex revealed a weak but significant difference in male LWR slope between lakes ( $p < 0.05$ ), and a highly significant difference in female LWR slope between lakes ( $p < 0.0001$ ). Fish in forested lakes matured earlier than those found in mixed vegetation and grass-vegetated lakes. Males were consistently smaller than females in all lakes ( $p < 0.0001$ ) but there were no differences in sex ratio. Sexually identifiable fish found in the forested lakes fell within the 110-185 mm size range, whilst most in the mixed and grassed floodplain lakes were within the 140 to >200 mm size range, with females being larger and more abundant in the grassed floodplain habitats and with distance from the mine. In all the lakes, the majority of the individuals and adults were found in the 110-215 mm size classes, comprising 84.1% of the total population sampled. The juveniles from 35-85 mm made up 13.7 %, and those in size classes greater than 215 mm were 2.2% of the population. There were no

clear seasonal differences in size classes; however, there were significant differences in size-class distributions between the lakes.

The study of growth of this species was undertaken by tracking cohorts in wild population, while aging of this species in the wild was not possible by using otoliths, where 37 otolith samples showed between 3 to 14 checks. The tracking of cohorts in the lakes showed overlapping cohorts year round clearly indicating continuous spawning. Data from pond fish showed that it is not possible to use otolith or scale checks to estimate the age of *N. papuensis* as several checks are formed on otoliths every year, not corresponding to fixed time periods. The study showed that *N. papuensis* is a fast-growing species with a mean growth rate in the Fly River lakes of 13.69 mm/month, whilst that of fish grown in ponds was 11.86 mm/month. A Von Bertalanffy fitted age-length equation of  $L(t) = 434.73 [-\exp(-2.43604*(t-0))]$ , was derived for pond-raised *N. papuensis* the Fly River system.

Understanding the diet of species which play a key role in an ecosystem is important because of its links to higher order consumers. This aspect of the study was undertaken by studying gizzard contents as well as stable isotopes. A total of 280 gizzard samples (140 during wet and 140 during dry season) were collected of which 202 had food while 78 were empty, and 280 muscle tissue samples were collected for stable isotope analysis. *N. papuensis* was shown to be predominantly planktivorous and detritivorous through all seasons, however its gut contents and fullness is habitat and season dependant ( $p < 0.0001$ ) with gizzards being fuller in the wet than in the dry season. The identifiable microalgae in the gizzard comprised 32 genera from the Chlorophyta (50%), Bacillariophyta (25%), Euglenophyta (9.4%), Cyanophyta (6.3%), and Dinophyta, Cryptophyta and Chrysophyta (3.1% each). The largest percentages of phytoplankton were found in fish from the grassed floodplain lakes while detritus content was higher in fish from forested lakes. Results showed clear interaction between lake, season, and food category ( $p < 0.0001$ ). The stable isotope results showed that fish from the grassed floodplain lakes were more  $\delta^{13}\text{C}$  enriched than those from the forested lakes ( $p = 0.0001$ ). There was also a significant difference in  $\delta^{15}\text{N}$  between lakes ( $p < 0.001$ ). There was a clear transition between the carbon signatures of fish from forested (very depleted  $\delta^{13}\text{C}$  values close to those typical of C3 plants) to the strictly grassed floodplain lakes (relatively enriched  $\delta^{13}\text{C}$  values close to those typical of C4 plants). Fish from lakes that have a mixed riparian vegetation of forests and open grass - aquatic macrophytes had  $\delta^{13}\text{C}$  values in the intermediate range. Generally, *N. papuensis* had C and N signatures that were closer to phytoplankton, periphyton, zooplankton and particulate organic matter.

Maturity stages were present in *N. papuensis* populations across a range of sizes, and throughout the year. In general, where both males and females were found in either season, stages 3 and 4, the most fertile stages, were the most common, and occurred more frequently in the grass floodplain lakes than in the forested reaches. *N. papuensis* was shown to be a highly fecund species with  $22,688 \pm 5,033$  to

126,812 ± 14,916 eggs per fish, and with oocyte diameters ranging from 0.41 ± 0.05 mm to 0.58 ± 0.01 mm. There was considerable variation in male and female GSI in all lakes and between seasons despite *N. papuensis* being a year-round spawner. Female GSI in these lakes fell within a range of 1.31 ± 0.20 to 4.38 ± 0.30, and male GSI was within a range of 0.59 ± 0.08 to 2.47 ± 0.21. There was strong evidence that higher levels of Cu in the mine-impacted lakes have negatively affected the reproductive ability of *N. papuensis*. The results suggested that with increase in dCu in the water column there is a corresponding decrease in GSI. This relationship was observed in both male and female fish.

This study is only the second species-specific study of the life history of Fly River fishes, despite there being over 120 different fish and macroinvertebrate species in this river system – the largest wetland system in Australasia and the south Pacific – and despite this species being a keystone species as food for predatory fish such as barramundi. However, of greatest concern is that the future of this system is under serious threat from mining and climate change. There is a serious need to research further the implications of these issues on fisheries ecology, proposed fishery developments, and community livelihood.

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## Glossary of Acronyms

ANZECC/ARMCANZ	Australian and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand 2000, <i>Australian Water Quality Guidelines for Fresh and Marine Waters, National Water Quality Management Strategy</i> , Australian and New Zealand Environment and Conservation Council, Canberra. <a href="http://erin.gov.au/water/quality/nwqms/index.html">http://erin.gov.au/water/quality/nwqms/index.html</a>
ANCOVA	Analysis of Co-variance – a combination of analysis of variance (ANOVA) and linear regression that accounts for intergroup variance when performing ANOVA. Including a continuous variable (the covariate) in an ANOVA model will account for known variance not related to the treatment.
ARD	refers to the acidic water that is created when sulphide minerals are exposed to air and water and, through a natural chemical reaction, produce sulphuric acid.
BOS10	Bosset Lagoon, a large blocked valley lake in transition zone between forested and grassed floodplain systems.
BVL	Blocked Valley Lake, Lake formed at the back of floodplains as a result of buildup of levee banks.
dCd	Dissolved cadmium, elemental cadmium in dissolved form.
dCu	Dissolved copper, elemental copper in dissolved form.
DO	Dissolved oxygen, oxygen dissolved in water and available to aquatic organisms for respiration.
dPb	Dissolved lead, elemental lead in dissolved form.
dZn	Dissolved zinc, elemental zinc in dissolved form.
DAV01	Lake Daviambu, a blocked valley lake.
<i>El Niño</i>	is the warm phase of the <i>El Niño</i> Southern Oscillation (commonly called ENSO) and is associated with a band of warm ocean water that develops in the central and east-central equatorial Pacific (between approximately the International Date Line and 120°W), including off the Pacific coast of South America.
ENSO	<i>El Niño</i> Southern Oscillation; refers to the cycle of warm and cold temperatures as measured by sea surface temperature, of the tropical central and eastern Pacific Ocean.
FPC	Flood Pulse Concept – theory that proposes that the annual flood pulse is the most important aspect and the most biologically productive feature of a river's ecosystem. The flood pulse concept describes the movement, distribution and quality of water in river ecosystems and the dynamic interaction in the transition zone between water and land.
GFI	Gut Fullness Index; the ratio of gut contents weight to body weight as: $GFI = [\text{Weight of gut contents (g)} / \text{Weight of Fish (g)}] \times 100$ .
GFL	Grassed Floodplain Lake, either grassed or forested lake in the floodplain.
GSI	Gonado-somatic Index, is the calculation of the gonad mass as a proportion of the total body mass.



IPM	Inorganic Particulate Matter – particulate matter in water column or sediments not of plant or animal matter.
<i>La Nina</i>	is a coupled ocean-atmosphere phenomenon that is the counterpart of <i>El Niño</i> as part of the broader <i>El Niño</i> –Southern Oscillation climate pattern. It brings with it higher than average rainfall.
LCF	Length at caudal fork – fish length measured to the edge of the caudal fork.
LEV01	Levame Lake (on the Strickland River).
LWR	Length-Weight Ratio – as fish grow in length, they increase in weight. The relationship between weight and length is not linear. The relationship between length ( <i>L</i> ) and weight ( <i>W</i> ) can be expressed as: $W=aL^b$ .
NTU	Nephelometric Turbidity Unit – a unit of measurement of the cloudiness or haziness of a fluid caused by suspended solids.
OBL	Oxbow Lake – horse-shoe shaped lake formed from cut-off meander loops.
ORWB	Off river water body – all grassed floodplain and forested lakes.
OTML	Ok Tedi Mining Limited – the copper/gold mining company at the headwaters of the Ok Tedi River, previously owned by BHP Billiton.
AXB01	Oxbow 1 – forested oxbow near the township of Kiunga, away from mining effluent.
AXB02	Oxbow 2 – forested oxbow lake immediately downstream of the Ok Tedi-Fly River confluence which has been largely filled with sediment.
AXB06	Oxbow 6 – forested oxbow lake at the end of the forested reach of the Fly, which is recipient of a large amount of mining effluent.
AXB05	Lake Pangua, an oxbow lake.
OPM	Other Plant Matter.
PNG	Papua New Guinea.
PCA	Principal Component Analysis – an ordination technique, used to display the relative positions of points in multivariate space in fewer dimensions, while retaining as much information as possible. It is essentially an exploratory technique that allows investigation of patterns of variation and relationships among the points.
POM	Particulate Organic Matter – material of plant or animal origin that is suspended in water, or deposited on sediments.
RCC	River Continuum Concept – a model that states that a river’s biological and chemical processes correspond to its physical attributes, and that the nature of biological communities changes in a downstream direction just as the river itself does. This means that the structure of the biological communities is also predictable and reflects the particular conditions of a stretch of stream.
RPM	River Productivity Model – this model emphasises the importance of local in-stream production (e.g., phytoplankton and benthic algae), and direct inputs of organic matter from the adjacent riparian zone.
TSS	Total suspended solids – solids in the water that can be trapped by a filter. TSS can include a wide variety of material, such as silt, decaying plant and animal matter, industrial wastes, and sewage. High concentrations of suspended solids can cause many problems for stream health and aquatic life
UOM	Unidentified Organic Matter

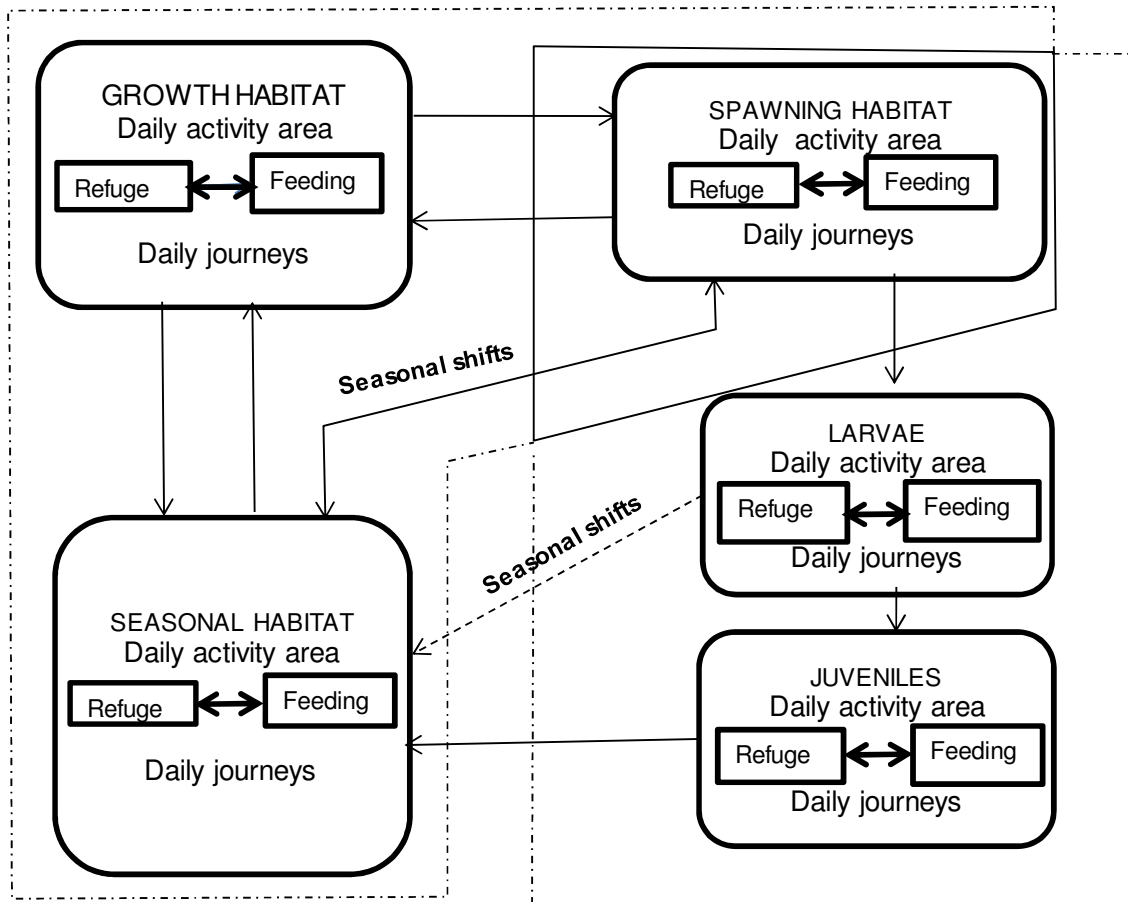
## **Chapter 1 Introduction**

### **1.1 Introduction**

Understanding fish biology and ecology is vital in understanding the environmental issues confronting freshwater lake and river ecosystems, subject to increasing pressure from artisanal and commercial fisheries, as well as from resource development projects such as mining. This is particularly true in large tropical floodplain rivers with high population densities and rapid natural resource extraction and development (e.g. Tockner & Stanford, 2002; Adeyamo, 2003), and where natural fisheries form a critical staple for local people (Lowe-McConnell, 1996; Tockner & Stanford, 2002; Coomes *et al*, 2010).

In many of these systems the major environmental issue is habitat destruction or alteration (Roni *et al*, 2005), which culminates in a range of impacts on fish at the cellular, organ, individual, population and eventually community levels of organisation (Lloyd, 1992; Lawrence & Hemmingway, 2003; Newman & Unger, 2003). Responses to changes in habitat condition and quality over a variety of spatial and temporal scales, at the population and/or community level of organisation, are usually a consequence of chronic exposure to impacts at the lower levels of biological organisation (Lawrence & Hemmingway, 2003; Newman & Unger, 2003). Such changes affect the ecology of aquatic systems and consequently the structure, dynamics and other biological functions of aquatic organisms.

Aquatics habitats are defined by their structure, which includes water depth, substrate type, aquatic vegetation and bank cover, current and occurrence of woody debris, water depth and velocity, connectivity and occurrence of pools, backwaters, and gravel runs, and by water physico-chemistry (Bishop & Harland, 1982; Bishop & Forbes, 1991; Sheldon & Meffe, 1995; Tejerina-Garro *et al*, 1998; Cowx & Welcomme, 1998; Buffagni *et al*, 2000; Araoye, 2009; de Melo *et al*, 2009). Fish community structure, dynamics, diet, reproduction, growth, and movement are influenced by the quality of the habitat, and fish require suitable habitats for their different life stages (Figure 1.1). Fish move longitudinally upstream and/or downstream, or laterally to backwaters, vegetated margins, flooded oxbow lakes, lagoons, and floodplains, in search of spawning, feeding, rearing/nursery and predator-avoidance areas. Some employ lateral movement pathways to avoid harsh conditions such as flooding, or to enable them to remain in shelter until they are ready to face stage-specific challenges. Lateral migrations also serve a critical role in resupply of fish assemblages after extinctions due to adverse conditions (Smith & Bakowa, 1993; Sheaves & Johnston 2008). Vertical movements involve fishes' use of particular parts of the water column for specific functions; for instance, the use of river or lake beds/sediments for spawning and burial of eggs and for its survival, which is influenced by factors such as dissolved oxygen concentration, particle size and water velocity.



**Figure 1.1 Fish activities in different habitats during different life stages, where clear movements occur between adult populations, and also to spawning grounds and back. There is only a single direction of movement between spawning grounds to larvae, juveniles, and into the population (Modified from Welcomme & Cowx, 1998)**

Water quality is critical in determining the integrity of the world's freshwater wetlands and marine environments. It is affected by the climatic conditions and natural physicochemical properties of the water body and their interactions with pollutants associated with anthropogenic activities such as changes in land use and industrial development, the effects of which on aquatic flora and fauna are well documented (e.g. Kadlec, 2003; Thomas *et al*, 2003; Kantawanichkul & Neamkam, 2003).

When fish and other organisms experience environmental disturbances, which lie outside of the normal ambient range, the effects can be dramatic (Jobling, 1995). A range of contaminants, such as nutrients, metals and hydrocarbons can adversely affect individuals and have long-term implications for the ecosystem. In many cases, contaminants have compounding effects, and/or can only have an impact in the presence of another contaminant and under certain prevailing physicochemical regimes (Lawrence & Hemingway, 2003). These impacts can be manifested at the lower levels of biological organisation (cellular level) and progress to more complex levels of organisation (population and community responses) (Lloyd, 1992). This seems to be the normal progression because impacts seen at the higher levels of organisation are thought to be the corollary of long-term exposure to water

quality degradation at sub-lethal levels at the lower levels of biological organisation (Lawrence & Hemingway, 2003; Newman & Unger, 2003).

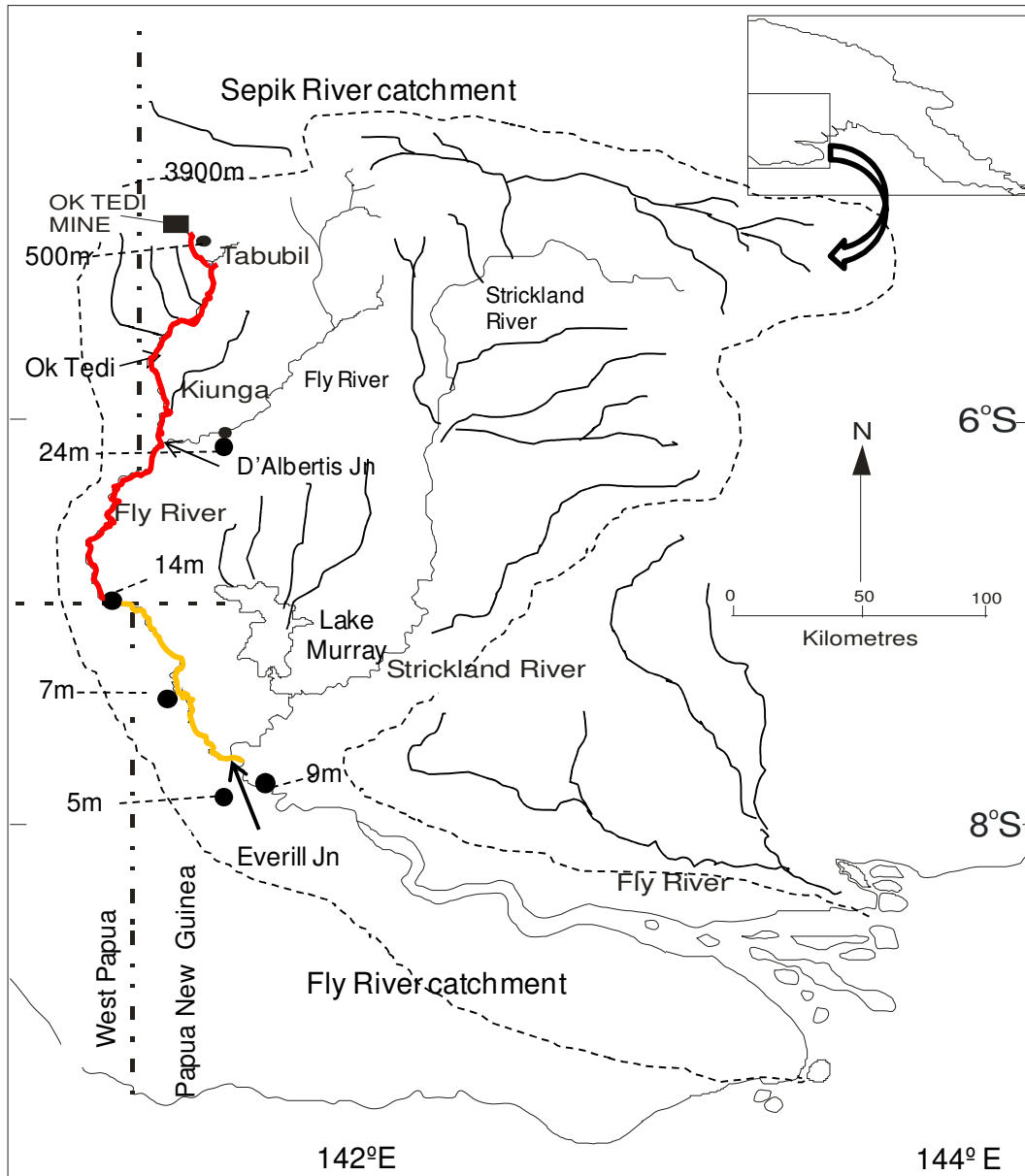
The extreme plight of the tropical aquatic environment is captured by Adeyamo (2003), who states that fish and marine resources in Nigeria face total collapse or extinction due to over-fishing and destruction of aquatic life and natural habitats by the pollution of water bodies. Furthermore, he added that the unregulated and excessive use of pesticides for fishing and the deliberate disposal, and dumping of toxic and hazardous wastes into water bodies, have caused massive fish kills and loss of aquatic life and habitats in that country. The plight of the Nigerian waterways appears to be comparable to the large Fly River system of Papua New Guinea (PNG), where mining activities are negatively impacting on its flora and fauna (Smith & Morris, 1992; Swales *et al*, 1998; Swales *et al*, 1999; Storey *et al*, 2009a, b; Bolton *et al*, 2009; Pickup & Marshall, 2009)

Seasonal changes in habitat and water quality have a major bearing on biological functions, ecology and distribution. Changes to habitat conditions result in fish seeking out different habitats/microhabitats that provide the necessary resources to support the needs of particular life-history stages (e.g. Rodriguez & Lewis, 1997; Cowx & Welcomme, 1998; Arthington *et al*, 2005; Rolls & Wilson, 2010).

The complex interaction between biology and impacts means that we need an extensive understanding of the biophysical factors that govern fishes' biological activities in their natural habitat. The aim would be to put in place appropriate management plans for large tropical rivers and wetland systems. This chapter points out the lack of necessary data and, therefore, suitable management plans, using the Fly River herring, *Nematalosa papuensis* as a case study. This species was specifically targeted for study since it is considered to be a keystone species in the Fly River floodplain.

## **1.2 The Fly River system**

The Fly River rises at approximately 3,000 m above sea level in the karst Star Mountains and flows 1,200 km to the Gulf of Papua. The catchment area is 76,000km<sup>2</sup> and includes three major tributaries: the Ok Tedi (*Ok* means river in the local Faiwol and Yonggom languages) drains the Hindenberg Ranges; the Upper Fly drains the southern part of the Victor Emmanuel Range; and the Strickland drains the Victor Emmanuel and Central Ranges (Pickup & Marshall, 2009) (Figure 1.2). The Fly's total mean discharge is 6,000 m<sup>3</sup> s<sup>-1</sup>, equivalent to the Danube, Niger or Zambesi (Welcomme, 1985). The Fly is ranked 23<sup>rd</sup> in the world in terms of its discharge and outranks many of the world's major rivers in terms of run-off per unit catchment area, which is only 76,000 km<sup>2</sup>. It has a very extensive, seasonally inundated floodplain; its scale is evident from the fact that the location of the river port of Kiunga, 800 km from the Gulf of Papua, is only 24 m above sea level.



**Figure 1.2 Catchment and drainages of the Fly River system within dotted lines. The Sepik River drains the north of this catchment. Red line shows river reaches with serious river bed aggradation and forest dieback. Yellow indicate river reach with serious river bed aggradation.**

The Fly is the largest river on the island of New Guinea and feeds the largest wetland system in the South Pacific, including Indonesia and Australia (Ellison, 2009). The main features of this floodplain system include oxbow lakes (OBL), blocked valley lakes (BVL), and grassed floodplain lakes (GFL) (Smith & Bakowa, 1993). Oxbow lakes are formed from meander cut-offs from the river channel. These are usually deeper than 10 meters and ‘U’-shaped in bathymetry. The BVLs are formed by the hydraulic damming up of floodplain waters by river channel and levees. These are shallow water bodies, usually < 7 m deep. The GFLs are very temporary and shallow water bodies (<3 m deep) that are flooded with every rise in river level and drained as the river levels drop.

This river system, like other major river systems in the tropics, is under increasing stress from the joint threats of population pressure and large-scale mining and other resource developments. In particular, it is currently under extreme pressure from pollution by mine wastes from the large open-cut Ok Tedi gold/copper mine. Various studies (e.g. Storey *et al*, 2009a; Pickup, 2009; Pickup & Marshall, 2009) have shown that there has been a continuous buildup of sediment in the Fly River bed over time as a consequence of the dumping of tailings and waste rock into the river system from the operations of the mine. This riverbed aggradation produces prolonged overbank flooding and sediment deposition, resulting in large areas of forest die-back (Pickup & Marshall, 2009; Storey *et al*, 2009a; pers. obs.). Long-term predictions indicate that these impacts will continue long after mine closure, as the sediment front progresses downstream (Pickup, 2009). These impacts will alter the hydrological regime as well as the chemical and biological processes that govern the ecology of riverine ecosystems and off-river water bodies (Higgins, 1990).

Although the changes and pressures in large tropical river systems are likely to be similar to those in temperate systems, little study has been undertaken to verify this or to determine how the outcomes of these pressures might be altered in a hot and humid tropical setting, or in the face of diverse tropical fauna and complex food webs. Consequently, the implications of various impact scenarios on key ecosystem components are unknown. In particular, ecological threats to key species, such as phytodetrivores, which feed at the base of the food chain, are poorly understood (e.g. Whitehead, 1985; Nelson, 2006; FAO, 2007). Similarly, their responses to environmental changes are unknown, even though this is likely to offer insights into how aquatic food webs will be impacted and so allow robust predictions and the development of responsible management plans (e.g. Sirimongkonthaworn & Fernando, 1994). It is therefore vital for ecological and socioeconomic reasons to understand in some detail the background and ecology of apparently key species, such as *Nematolosa papuensis*, the subject of this thesis.

### **1.3 Fisheries ecology of the Fly system**

The fish fauna of the water bodies of the Fly River floodplain occur in the three main types of off-river water bodies described above. Fish diversity and abundance is more consistent in OBLs than in BVLs and GFLs (Storey, 2005; Hortle & Storey, 2006), and there are significant inter-annual variations in fish biomass in BVLs and GFLs, with notable declines in diversity and abundance recorded during extreme weather conditions such as the *El Niños* (Swales *et al*, 1999; Storey *et al*, 2009b). These periods of extreme droughts result in very low river levels, with most habitats on the floodplain drying up (Swales *et al*, 1999). Oxbow lakes are the exception; being deeper (generally 10–20m deep), they continue to retain water, but may also have contributions from the water table, and therefore provide a permanent drought refuge for fish (Smith & Bakowa, 1993; Swales *et al*, 1999). Differences in fish communities among the different floodplain habitats reflect differences in habitat

structure and the different effects of drying and flooding (e.g. Swales *et al*, 1999; Mol *et al*, 2000). Nevertheless, this system supports a diverse and abundant suite of habitats and assemblage of freshwater fish, with 108 species being recorded (Allen *et al.*, 2008); this is twice the number of species recorded in the contiguous Sepik River (Allen & Coates, 1990), the major drainage system flowing to the northeast coast of PNG.

Fly River fish assemblages have continually shown significant declines in catches (biomass and numbers of fish) at the majority of riverine sites downstream of the mine (OTML, 1994, 1995, 1996; Swales *et al*, 1999; Storey *et al* 2009a, b) with the extent of declines reducing in severity with increasing distance from the mine. Two issues stand out with respect to the current plight of Fly River fish fauna. Firstly, no detailed study has yet been undertaken to determine different off river water body (ORWB) habitat conditions and the variables that govern fish ecology in these lakes. Secondly, fish assemblages in the Fly River system have always been described at the community level, and only one assessment has ever been made in relation to impacts of mining wastes on an individual species – the barramundi (*Lates calcarifer*) – with no conclusive evidence of negative impacts of mining on its biology (Milton *et al*, 2000). Studies at the species level are important, particularly for ‘keystone species’, whose loss would precipitate many further severe population declines or extinctions (Mills *et al*, 1993). Moreover, detection of impacts on such species can give early warning of problems before they are apparent at assemblage levels.

#### **1.4 Study species**

The Fly River herring (*N. papuensis*) is a member of the gizzard shads (Clupeidae: Dorosomatinae), a group common in coastal shallows and estuaries throughout the subtropics (Miller 1960; Nelson & Rotham 1973) and in lentic fresh waters over much of North America (Miller 1960; Noble 1981). Its closest relative in the south Pacific region is the Australian bony bream, *Nematalosa erebi*, which is arguably the most widespread of all the freshwater species of Australia. *N. erebi* is found in waterholes in all of the larger rivers and in many of the smaller, more ephemeral tributaries in Australia. *Nematalosa papuensis* and *N. erebi* appear to fill a very similar ecological niche to their North American relative *Dorosoma* spp (Schaus & Vanni, 2000; Vanni *et al*, 2006; Zeug *et al*, 2009).

Whitehead (1985) listed nine species of *Nematalosa* in this region with a wide distribution. The *N. arabica* is found throughout the northwestern Indian Ocean; *N. nasus*, from northern Indian Ocean to the South China Sea; *N. galathea* from India to Viet Nam; *N. japonica*, Japan to the island of Taiwan. *N. come* has a distribution that extends from Australia to East China Sea; whilst *N. erebi* is arguably the most widespread inland freshwater species in Australia. The anadromous Perth herring, *N. vlaminghi*, is found only in Western Australia.

Two species were described from PNG's Fly River system; *N. flyensis* and *N. papuensis*, but there are doubts as to whether the former is really a different species (Whitehead, 1985). Furthermore, the results of a genetic investigation by Watts (1997) could not distinguish these two. It has to be noted that *Nematalosa* from the Fly system are quite variable morphologically: some fish exhibit extremes of the range for some characters, but fish are found with intermediates for those characters, including the main distinguishing characters as shown in Table 1. It is likely that the few specimens used to describe *N. flyensis* were individuals from one extreme of the morphological range, so they appeared to be distinct from those fish on which the description of *N. papuensis* was based at the other end of the range. Based on these arguments, it would appear to treat *Nematalosa* from the Fly River system should as a single species - *N. papuensis* - as they are in this thesis.

*Nematalosa papuensis* is very similar to the Australian species *N. erebi* with which it shares a recent common ancestry, indeed on morphological grounds it is difficult to see why these species should be regarded as distinct, as their main diagnostic features overlap (Table 1.1). Note that *Nematalosa* recorded from the Bensbach and Digoel Rivers; southwest of the Fly have been reported as *N. erebi* (Hitchcock, 2002). Regardless of this taxonomic issue, should a fishery based on *Nematalosa* be established, it would be impossible to design gears to select one or other of the 'species'.

**Table 1.1 Main morphological features of the three described freshwater *Nematalosa* species from Australasia (Allen, 1991; Allen *et al*, 2002; www.fishbase.org)**

Character	Described species		
	<i>N. erebi</i>	<i>N. papuensis</i>	<i>N. flyensis</i>
Dorsal Rays	14-19	16-17	13-14
Anal Rays	17-27	23-24	17-26
Pectoral Rays	14-18	14	14-16
Pelvic Rays	8	8	8
Scutes	25-31	26-31	26-32
Gill Rakers	nd	160-320	360-520
Ratio of gill raker length to gill filament length	nd	>2/3	1/5-2/3

Studies in tropical Australia reveal that *N. erebi* is most commonly associated with still-water environments and is one of the few Australian fishes to be primarily herbivorous when mature (Bishop *et al*, 2001). Its diet consists of mainly algae and other plant material, although some crustaceans and insects are also ingested. They can survive a wide range of temperatures, between 9 and 38°C, and pH values from 4.8 to 8.6 (Merrick & Schmida, 1984). Annual kills are common



during winter, as a result of achlya infections (Puckridge *et al*, 1989), parasitic infections (Langdon *et al*, 1985) and low water temperatures (Merrick & Schmida 1984). Other details of the life history of *N. erebi* have been reported by Puckridge & Walker (1990), Arthington *et al* (1992), and Bishop *et al* (2001). In contrast, relatively little is known about most of the biology of *N. papuensis* (Table 1.1).

**Table 1.2 Available literature on *N.erebi* and *N.papuensis*. NI = No information available**

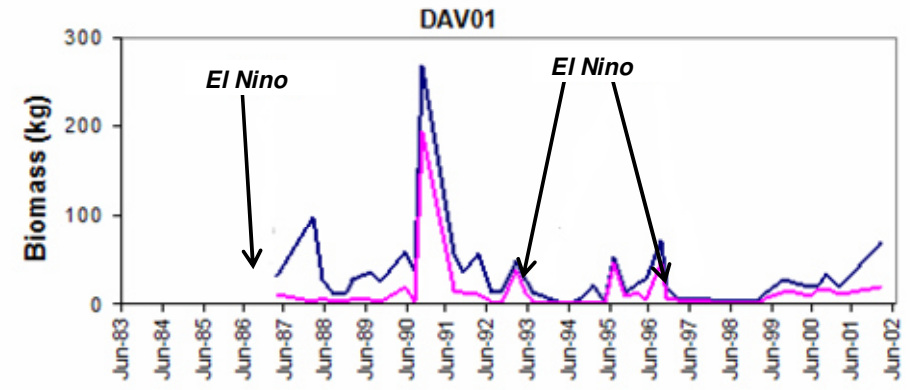
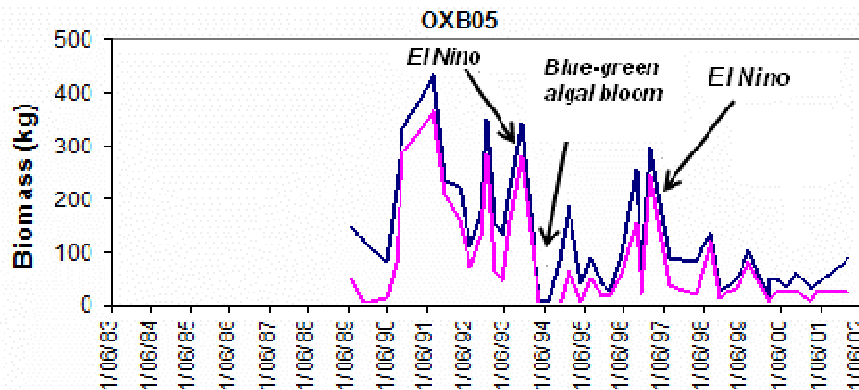
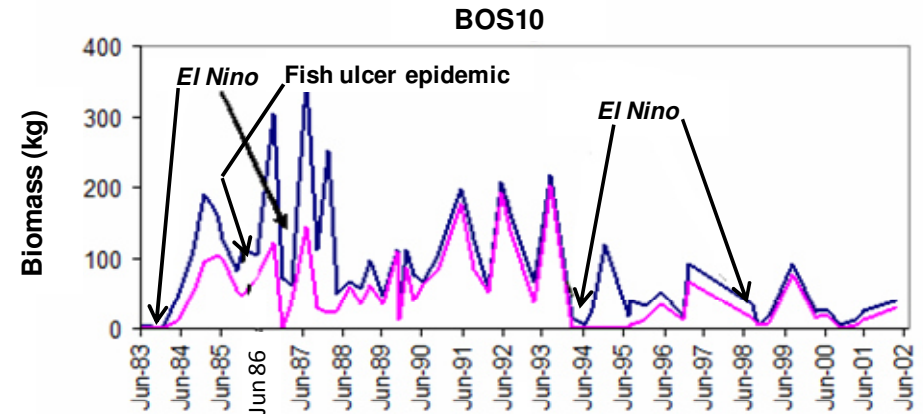
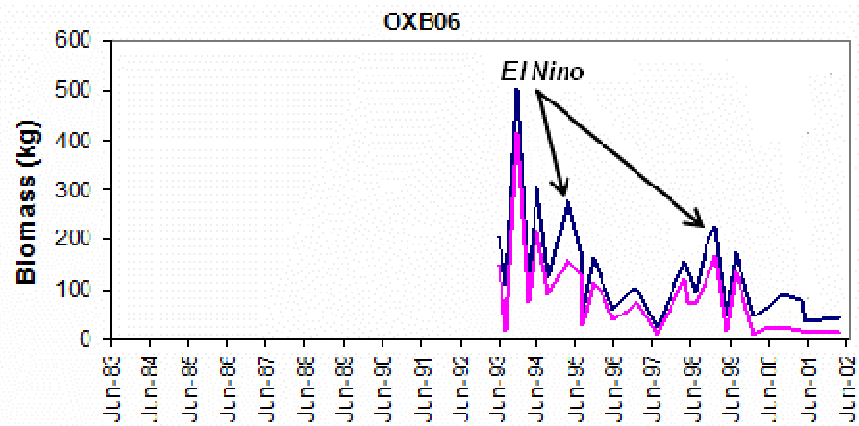
<b>Biological aspect</b>	<b><i>N.erebi</i> (Australia)</b>	<b><i>N.papuensis</i> (PNG)</b>
<b>Habitat</b>	<b>Billabongs, river and stream channels</b> (Bishop <i>et al</i> , 2001; Growns, 2004; Pusey <i>et al</i> , 2004)	<b>Off-river water bodies, Pelagic</b> (Smith, 1998)
<b>Diet</b>	<b>Algae, detritus, zooplankton</b> (Pollard ,1974; Pusey <i>et al</i> , 1995; Bishop <i>et al</i> ; 2001; Balcombe <i>et al</i> , 2005; Medeiros & Arthington; 2008; Sternberg <i>et al</i> ,2008)	<b>Algae, zooplankton</b> (Smith, 1998; Bunn <i>et al</i> , 1998; Bowles <i>et al</i> , 2001; Storey & Yarrao, 2009)
<b>Growth/Reproduction</b>	<b>Seasonal spawner, high fecundity, growth:</b> (Puckridge & Walker, 1990; Bishop <i>et al</i> , 2001; Growns, 2004)	<b>Maturity stages all year, high fecundity, fast growth rate</b> (Hortle & Storey, 2006)
<b>Population dynamics</b>	<b>Habitat preference, seasonal changes</b> (Bishop <i>et al</i> , 2001; Growns, 2004)	NI
<b>Population structure</b>	<b>More females; Females bigger; seasonal changes in distribution</b> (Puckridge & Walker, 1990; Bishop <i>et al</i> , 2001)	<b>More females than males in grass land lakes, Females bigger</b> (Hortle & Storey, 2006)
<b>Importance and susceptibility to environmental changes and human impacts</b>	<b>Susceptible to low flow/low temperatures in winter</b> (Langdon <i>et al</i> , 1985; Gehrke & Harris, 2001; Puckridge <i>et al</i> , 2001); <b>Pesticide residues indicator</b> (Kumar & Chapman, 2001)	NI

In both abundance and biomass, *N. papuensis* is the most common fish in the Fly River system; both in the river channel and ORWBs. This is similar to a number of other lacustrine systems in the tropics where clupeids are the dominant pelagic taxon (Fernando & Holčík, 1982). Information on this species' population and size structure in some ORWBs have been briefly mentioned in two publications (Smith & Bakowa, 1993; Storey *et al*, 2009b), one Ok Tedi Mining (OTML) biology annual report (OTML, 1996) and some fisheries development proposals and studies (Wilson, 1992; Hortle & Storey, 2006). Smith & Bakowa (1993) estimated this species to be 14-72% of the ORWB fish biomass in the forested and grassed floodplain lakes. Recent reports (OTML, 1996; Storey *et al*, 2009) placed the estimated biomass of this species at approximately 37-38% of all the ORWB fish

catches. Such abundance indicates that this species is very dominant and is probably a key driver of the fisheries ecology of this river system.

The position of *N. papuensis* as a major phytodetrivore/phyto-planktivore in the Fly river system (Smith, 1998; Bunn *et al*, 1999; Bowles *et al*, 2001; Storey & Yarrao, 2009) means it is responsible for converting large amounts of plant carbon into fish biomass, therefore determining in large part the material and energy flows from the phytodetritus nutrient pool to higher trophic levels. With its numerical and biomass dominance, it might therefore be classified as a 'keystone' species according to the definition of that term by Mills *et al* (1993). Data in Hortle & Storey (2006), and OTML (unpublished) show that *N. papuensis* is a key driver of the fisheries ecology of the Fly River floodplains, as clearly shown in Figure 1.3, where fish catches (biomass) at lakes with long-term historical data show a strong influence of *N. papuensis* in the catches. Its role as a keystone species is strongly suggested by the fact that of the 66 species found in ORWBs, *N. papuensis* makes up to 38%, *Lates calcarifer* 7%, and all the others 55% of the catch biomass (OTML, 1996; Storey *et al*, 2009). This relationship between *N. papuensis* and overall fish catches, and the reported declines in catches downstream of the mine impacts, suggests the biology of *N. papuensis* may have also been significantly impacted by habitat degradation. However, the extent of impact of changes in habitat quality on *N. papuensis* are unknown, as are potential chain reactions propagated through the food webs in which *N. papuensis* plays a critical part.

This species may therefore represent a useful model for the development of clear understanding of the likely impacts of the complex of threats on both phytodetrivorous fish themselves and on the food webs in which they are keystone components. An understanding of such a species and its habitat requirements has implications for the management and conservation of fisheries in similar tropical systems in New Guinea and elsewhere.



— Total catch  
— *Nematalosa* only

Figure 1.3 Historical changes in *N. papuensis* populations in 4 lakes with the longest historical data sets. (Modified from Hortle & Storey, 2006)

## 1.5 Aims of this study

The structure and dynamics of riverine fish populations worldwide have been studied extensively and described under various river productivity models (Vannote *et al*, 1980; Schlosser, 1982; Junk *et al.*, 1989, Thorp & DeLong, 1994). Consequently, fish spatial and temporal dynamics are in part functions of habitat structure and conditions (e.g. Bishop & Harland, 1982; Schlosser, 1982; Harris & Kangas, 1988; Bishop & Forbes, 1991; Welcomme & Cowx, 1998; Buffagni *et al*, 2000). A number of useful models have been proposed to explain community dynamics and structures according to a range of spatial and temporal factors (Jackson *et al*, 2001), leading to a set of theories explaining community variations in rivers and streams, and patterns of productivity along lateral and longitudinal gradients (Vannote *et al*, 1980; Schlosser, 1982; Junk *et al*, 1989; Thorp & DeLong, 1994). Fish distribution and movements has been described in a four dimensional framework – i.e. longitudinal, lateral, vertical, and temporal (seasonal) – illustrating the dynamic spatial and temporal nature of fish movements and habitat requirements in freshwater systems. (e.g. Welcomme & Cowx, 1998; Silvano *et al*, 2000; Rolls & Wilson, 2010). Much of what is known about the large tropical floodplain systems of the world is the result of work undertaken in Amazonian, Asian and African wetlands (Vannote *et al*, 1980; Schlosser, 1982; Junk *et al*, 1989, Thorp & DeLong, 1994). Very little information, however, is available on floodplain fisheries ecology of the Indonesian archipelago, the island of New Guinea or the South Pacific. The intent of this study, therefore, is to determine the likely key drivers for the population structure, dynamics, and distribution of *Nematolosa papuensis*, a keystone phytodetrivore, under the current conditions of the Fly River system, and to establish an understanding of tropical freshwater ecosystems dominated by phytodetrivores.

In this study I describe the variability in key ecological features of populations of *N. papuensis* across a range of types of off-river water bodies in the Fly River system. I also assess the current natural and anthropogenic environmental impacts on this species and evaluate its vulnerability to impacts on this river system. I investigate whether this species is resilient to natural and anthropogenic pressures and whether its population dynamics clearly reflect habitat conditions in the Fly River lakes. It is envisaged that, firstly, knowledge of the dynamics of *N. papuensis* populations may present a useful model for understanding the spatial and temporal dynamics of fishes of large tropical lakes, and secondly, by understanding the biophysical nature of its environment and the current environmental impacts exerted upon it, I will be able to develop a model of the likely impacts of environmental threats on tropical lake food webs that are dominated by phytodetrivores.

To achieve these aims, the chapters of this thesis are structured to answer the following questions.

- I. Chapter 2: What are the current biophysical conditions of *N. papuensis* habitats in the Fly River system? This chapter describes the habitat, including water quality, in which *N. papuensis* is found. Current environmental impacts on the habitat by mining effluents are presented and discussed.
- II. Chapter 3: How do the dynamics of populations vary across different habitats? This chapter describes the spatial and temporal dynamics of this species over the last 20 years and during the 2 years of this study, and explores how mining impacts may have affected the fish assemblages.
- III. Chapter 4: What is the population structure of this species in the different habitats? Here I describe the population structure of *N. papuensis*, by analysis of the length-weight relationships of adults and juveniles in the different water body types. This chapter also examines the growth and age of this species and discusses how mine-derived pollutants may have already impacted the population structure of this species.
- IV. Chapter 5: How does this species' diet vary across different habitats? I address the dietary biology of the *N. papuensis* here by analysing gut contents of fish collected from the mine-impacted as well as non-mine-impacted lakes, and by using stable isotope analysis.
- V. Chapter 6: What are its spatial and temporal patterns of reproduction across different habitats? This chapter describes the reproductive characteristics of *N. papuensis* and in relation to habitat structure and quality.
- VI. Chapter 7: What does the understanding of its biology tell about the fishery ecology in this system, and how stable and resilient is it in the face of natural ecosystem variability and the impacts of human activities? This final chapter synthesises the results of this thesis.

## **Chapter 2 Biophysical conditions in off-river water bodies of the Fly River**

### **2.1 Introduction**

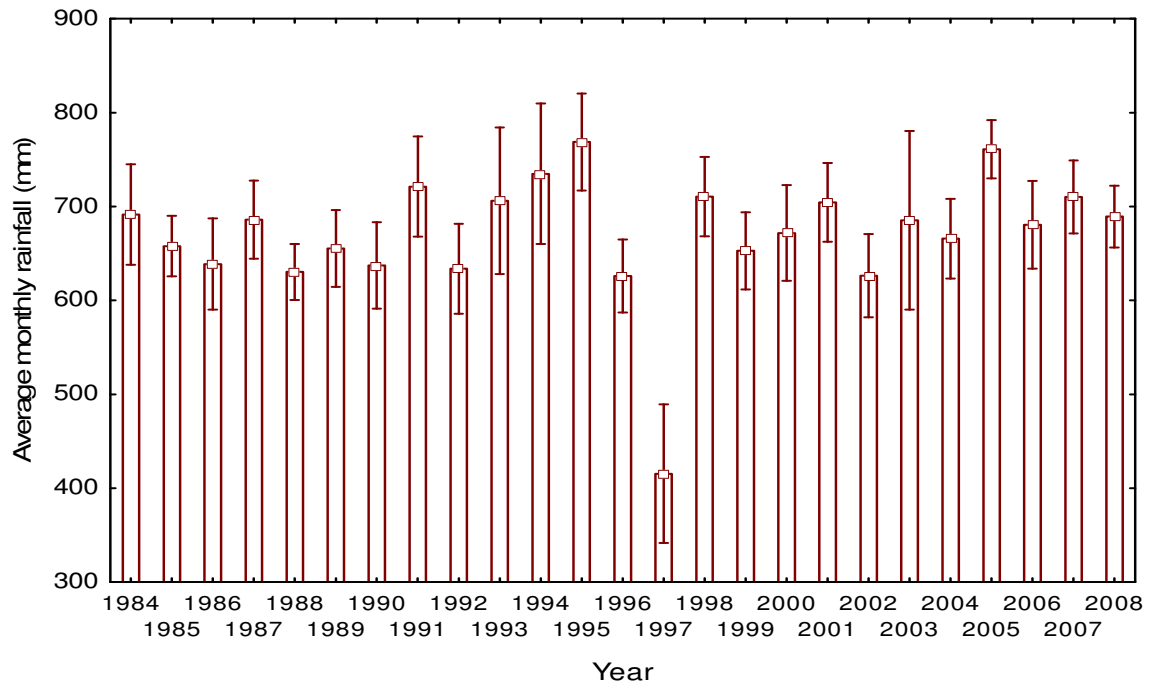
Describing the environment in which fish live can be difficult because of the many factors that influence their habitats both temporally and spatially (Winemiller *et al*, 2000). Suitable fish habitat should provide appropriate water quality, spawning grounds, areas for feeding and resting, refuges for avoidance of predators and environmental disturbances, and routes for migration (Harris & Kangas, 1988; Welcomme, 1998). The habitat should also be of a sufficient extent to cater for a viable population size. Habitat diversity is also required as many fish do not spend their entire life in the same habitat but seek microhabitat that is appropriate for each life stage by moving along longitudinal, lateral, and/or vertical pathways (Welcomme & Cowx, 1998). Various biophysical and hydrological factors that are known to influence fish community assemblages in the different habitats include altitude, dissolved oxygen concentration, pH, temperature, conductivity, current velocity, water depth, habitat heterogeneity, riparian vegetation cover, substrate type, snags and woody debris (Bishop & Harland, 1982; Cetra & Petrere, 2006; Araújo *et al.*, 2009, and Araoye, 2009).

*Nematalosa papuensis* is much more abundant in the off-river water bodies (ORWBs) than in the river channels of the Fly River (OTML unpublished data), hence the focus of this study on the ORWB habitats. The objective of this chapter, as listed in Chapter 1, is therefore to describe the current *N.papuensis* habitat, and to present the findings of this study in the context of the current broad-scale knowledge of the habitats and environmental conditions of Fly River ORWBs. This synthesis will provide a context for developing an understanding of the biology and ecology of *N. papuensis*, and of the likely human impacts on *N. papuensis* populations.

### **2.2 Description of the study area**

#### **2.2.1 Climate**

The climate of the Fly River region is tropical and humid with average rainfall varying with elevation (Moi *et al*, 2001). The rainfall in the upper catchment is in excess of 8,000 mm annually, with monthly averages of 600 – 800 mm (Figure 2.1), and is evenly distributed throughout the year. Total rainfall is about 8,000 mm in the upper and middle Ok Tedi, and about 1,800 mm in the floodplain reaches of the Middle Fly region (Pickup & Marshall, 2009). On the floodplain, the rainfall is monsoonally influenced, falling mainly in the wet season, from December to May. The region experiences major climatic cycles, such as the *El Niño* events of 1972, 1982/83, 1993/94 and 1997/98 (Pickup & Marshall, 2009). *El Niño* events cause severe droughts that last for several months as a result of warmer surface temperatures in the western Pacific region. The opposite of *El Niño*, *La Niña*, occurs when there is a state of unusually cool sea surface temperatures in the western Pacific, bringing with it wetter than normal conditions in PNG as well.



**Figure 2.1 Mean ( $\pm$ SE) monthly rainfall at the headwaters of the Ok Tedi at the mine site (Folomian weather station) from 1984-2008 (Data from Ok Tedi Mining Limited)**

### 2.2.2 Geomorphology and drainage

The upper tributaries of the Fly River (Figure 1.2) are fast flowing and capable of transporting large quantities of sediment, including boulders (Pickup & Marshall, 2009). The Ok Tedi and its tributaries drain the upper reaches across a heavily dissected landscape. The ridges rise to over 2,000 m in the north but most of the basin lies between 200 m and 800 m elevation. The eastern part of the basin is karst landform, including the massive Hindenburg Wall escarpment, and contains large areas of landslide debris and old debris flow deposits. The western part of the region has high mountains of igneous origin, but much of the area is of shale, limestone and sandstone origin. Much of the terrain is made up of unstable slopes despite the dense rainforest cover, with landslide and debris flows being a common occurrence (Pickup, 2001).

The operation of the Ok Tedi Mine discharges about 80 million tonnes a year of tailings and waste rock into the Mine Area Creeks and about 45 million tonnes of sediments, which includes sidewall erosion and landslide materials, from the Ok Tedi is transported into the Middle Fly region annually (Pickup, 2001). The monitoring of water flow at gauging stations and river cross-sectional surveys has been a major component of the hydrological monitoring program undertaken by the mining company, for over 20 years, along the full length of the Ok Tedi, Fly and Strickland River systems.

Extensive deposition of sediments in the lower Ok Tedi and the Middle Fly regions is now a major feature of the Fly River system since 1997 causing massive river bed aggradation, extending from Tabubil to Middle Fly region below the Everill (Fly-Strickland) Junction (Pickup, 2001, Pickup & Marshall, 2009). This has resulted in extensive dieback in forested areas and changes to more aquatic vegetation communities on parts of the lower floodplains (Pickup, 2009; Storey *et al*, 2009a).

The high altitude combined with high rainfall, seismic activity, and unstable geology results in naturally high sediment loads in the Fly and its major tributary, the Strickland River, giving an annual natural sediment discharge at its delta a very high sediment load of  $85 \times 10^6$  tonnes (ranked 17<sup>th</sup> in the world) (Walsh & Ridd, 2009). In fact, as Harris *et al* (1993) noted, the Fly River carries more sediments to the sea than all the rivers draining Australia combined.

The Strickland River, with a catchment area of 36,740 km<sup>2</sup> is much larger than the upper Fly (catchment of 18,400 km<sup>2</sup>) (Pickup & Marshall, 2009). Its higher sediment load is due to its closer proximity to the main sources of natural sediment from the central ranges. The gradient of the Strickland is steeper than the Fly and its floodplain is subjected to more extensive channel shifting. Waters backed up by the Strickland have formed some large off-river water bodies, including Lake Murray. The Strickland also has extensive terraces, 10-30 km wide, which were formed pre-Holocene and clearly separate the Fly and Strickland catchments (Pickup & Marshall, 2009).

### **2.2.3 Floodplain and hydrology**

The Fly River drains an area of about 75,000 km<sup>2</sup> in western Papua New Guinea. Three major rivers feed the large Fly-Strickland floodplain: the Ok Tedi which drains the Hindenburg Ranges; the Upper Fly which drains the southern part of the Victor Emanuel Range; and the Strickland which drains the Victor Emanuel and Central Ranges. The Upper Fly and the Ok Tedi meet at D'Albertis Junction to form the Middle Fly, which meanders down a 400 km long floodplain with extensive scroll bar complexes, oxbow lakes (OBLs) and blocked valley lakes (BVLs). The Middle Fly and the Strickland meet at Everill Junction before entering the Fly delta (Figure 1.2). The Fly delta covers an area of approximately 10,000 km<sup>2</sup> and extends downstream for another 400 km before entering the ocean in the Gulf of Papua. The Ok Tedi is the recipient of waste rock and tailings from the Ok Tedi mine while the Progera gold mine discharges its waste into the Strickland River.

The Fly has a very extensive, seasonally inundated floodplain; its scale is evident from the fact that the location of the river port of Kiunga, 800 km from the Gulf of Papua, is only 24 m above sea level, making it a very flat terrain. The hydrology of the Fly River system is influenced mainly by runoff that originates from the mountainous headwaters of its tributaries. The runoff just below the confluence of the Ok Tedi and Upper Fly River is reported to be 70-80% (6,600 mm/year) of the total rainfall, but this changes as it enters the floodplain reach of the middle Fly system (Pickup & Marshall, 2009). From the



confluence of the Ok Tedi and Fly to the top end of the floodplain reach there is considerable variation in water levels as flood waters enter the system; however, when these waters reach the floodplain, this short-term variability dissipates and the changes in water levels occur on a scale of weeks to months because of the water exchange between the floodplain water bodies and the river through tie-channels and levee breaches or directly across the floodplain during high floods (Pickup & Marshall, 2009). The “tie-channel” was a term coined by Blake & Ollier (1971) to describe the narrow channels that connect the main river channels and the OBLs on the Fly floodplain.

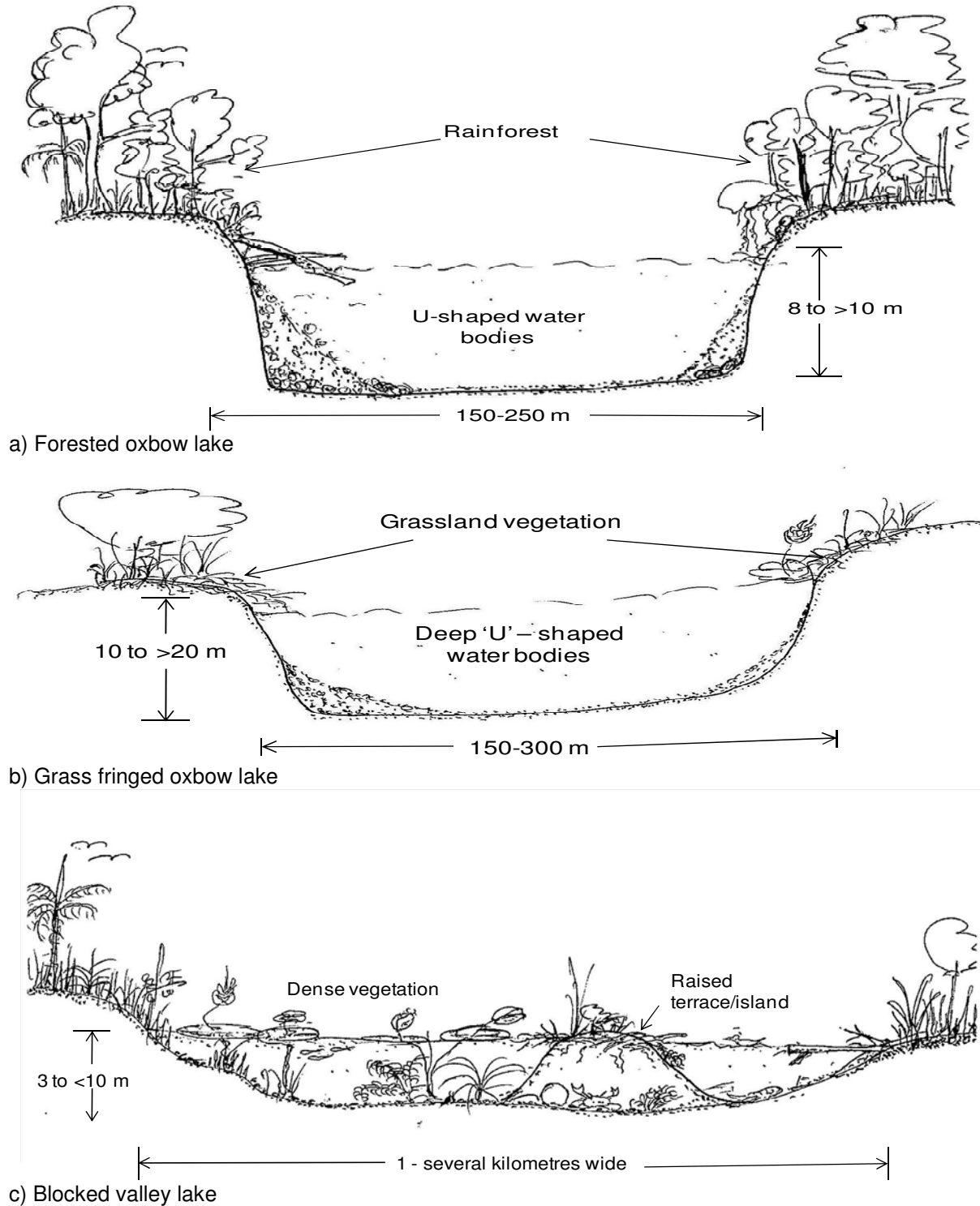
#### **2.2.4 Fly River wetland features**

This river system, like other major river systems in the tropics, is under increasing threat from population pressure and large-scale mining and other resource developments. In particular it is currently under extreme pressure from pollution by mine wastes from a large open-cut gold/copper mine. Various studies (e.g., Storey *et al*, 2009a; Pickup, 2009; Pickup & Marshall, 2009) have shown that there has been a continuous buildup of sediment in the Fly River bed as a consequence of the dumping of tailings and waste rock into the river system from the operations of the Ok Tedi Mine. This riverbed aggradation has produced prolonged overbank flooding and sediment deposition, resulting in large areas of forest die-back (Pickup & Marshall, 2009; Storey *et al*, 2009a; pers. obs.) stretching from the Ok Tedi into the Middle Fly below Everill Junction (Fly-Strickland Junction) (Figure 1.2). Long-term predictions indicate that these impacts will continue long after mine closure (initially set at 2013, but operations will now continue for another 10years), as the sediment front progresses downstream (Pickup, 2009). These impacts will alter the hydrological regime as well as the chemical and biological processes that govern the ecology of riverine ecosystems and off-river water bodies (Higgins, 1990).

The OBLs, BVLs and grassed and forested floodplain lakes are the prominent features of this floodplain. While the OBLs are more permanent, the BVLs can dry out completely during extreme drought conditions, and as noted previously, much of the grassed and forested floodplain dries each year for variable periods depending on the river level. Figure 2.2 and Plate 2.1 show the habitat types sampled during this study.

This large wetland comprises 122 km<sup>2</sup> of OBLs, 245 km<sup>2</sup> of BVLs, and 2473 km<sup>2</sup> of forested and grassed floodplain lakes. Oxbow lakes are formed from meander cut-offs from the river channel. These are usually deeper than 10 meters and U-shaped in bathymetry. The BVLs are formed by the hydraulic damming up of floodplain waters by river channel and levees. These are shallow water bodies, usually < 7 m deep. The GFLs are very temporary and shallow water bodies (<3 m deep) that are flooded with every rise in river level and drained as the river levels drop. Some of these lakes are

now under extreme pressure from the impacts of the operations of the Ok Tedi mine, such that lakes closest to the mine are heavily sedimented with mine-derived material.



**Figure 2.2 Cross-sectional view of the different habitats studied: a) forested oxbow, b) grass fringed oxbow and c) and blocked valley lake.**



**Plate 2.1** Typical off-river water body habitats sampled during this study. a) a grassed oxbow lake with a blocked valley lake in the background, b) a typical non mine-impacted forested oxbow lake, c) a typical blocked valley lake, d) a mixed forested and grass fringed oxbow lake, and e) a mine-impacted forested oxbow with dead trees in the background

## 2.3 Methods

### 2.3.1 Description and location of study sites

The locations of the study lakes and replicate sampling sites within the lakes are as shown in Figures 2.3 and 2.4. The BVLs and OBLs can be either forest-fringed lakes or grass-fringed lakes. The BVLs were formed where the Fly River acted as a hydraulic dam to flood broad shallow valleys, and OBLs were formed from cut-off meander loops of old river channels due to channel migration (Smith & Bakowa, 1993). The BVLs can be from one to several kilometers wide and several kilometers long with depths ranging from 3m to 10m. The oxbow lakes are more confined to the old river channels and can be from 150 to 300 m wide with lengths ranging from 1 to 5 km. Oxbow lakes are usually deeper than the BVLs, ranging from 8-10m in the forested reaches to greater than 20m in the floodplains. A general description of the study lakes is presented in Table 2.1.

The top-most study site on the Fly River (OXB01) is 40.7 km by river upstream of the confluence of the Ok Tedi and Fly Rivers and less than 1 km downstream of the copper shipment port of Kiunga. This is a forested oxbow lake surrounded by primary and secondary tropical forest vegetation.

The forested Erehta oxbow (OXB02) is located approximately 55 km by river downstream of the Ok Tedi-Fly confluence. Prior to mining at Ok Tedi, this lake was a forested oxbow, but following mine-derived sedimentation, forest die-back has caused it to become a more mixed-vegetation-fringed oxbow lake with the invasion of floodplain vegetation. For this study it is still classified as a forested oxbow.

The forested Kwem oxbow (OXB06) is located 183 km downstream of the Ok Tedi-Fly junction and is situated in the transition zone of the forested and floodplain reach of the Fly River system, which used to be surrounded by primary forest vegetation. It is located in the region which is experiencing the heaviest sedimentation impacts on both the terrestrial and aquatic flora and fauna. The forest dieback around this lake has resulted in the opening up of the canopy and therefore the establishment of terrestrial and aquatic grassed floodplain vegetation (Storey *et al*, 2009a). This oxbow is connected to a series of large blocked valley floodplain lakes to the east of the Fly River which are drained by the Agu and Binge Rivers.

Bosset Lagoon (BOS10), the largest of the study lakes, is a very broad but shallow body of water classified as a blocked valley lake. It is located at the edge of the lower end of the transition zone between the forested and grassed floodplain reach of the central Middle Fly region and has diverse aquatic vegetation. The northern to the southwestern edges of the lake are covered with rainforest vegetation while the eastern end is covered with grassed floodplain vegetation. It is classified as a grassed floodplain lake in this study.

Lake Pangua (OXB05), located approximately 394 km downstream of the Ok Tedi-Fly confluence and 13 km upstream of the Fly-Strickland junction (Everill Junction), is a complex double oxbow/blocked valley lake system, separated from Lake Daviambu (DAV01) by large growths of permanently

submerged and emergent aquatic grasses (Poaceae), water lily beds (approximately 300m across) and small islands. Lake Pangua is a deep lake (>20m) whilst Lake Daviambu is a very broad and shallow blocked valley lake (only 7 m deep during high flooding). Both of these lakes are true grassland lakes and are permanently connected by many tie channels.

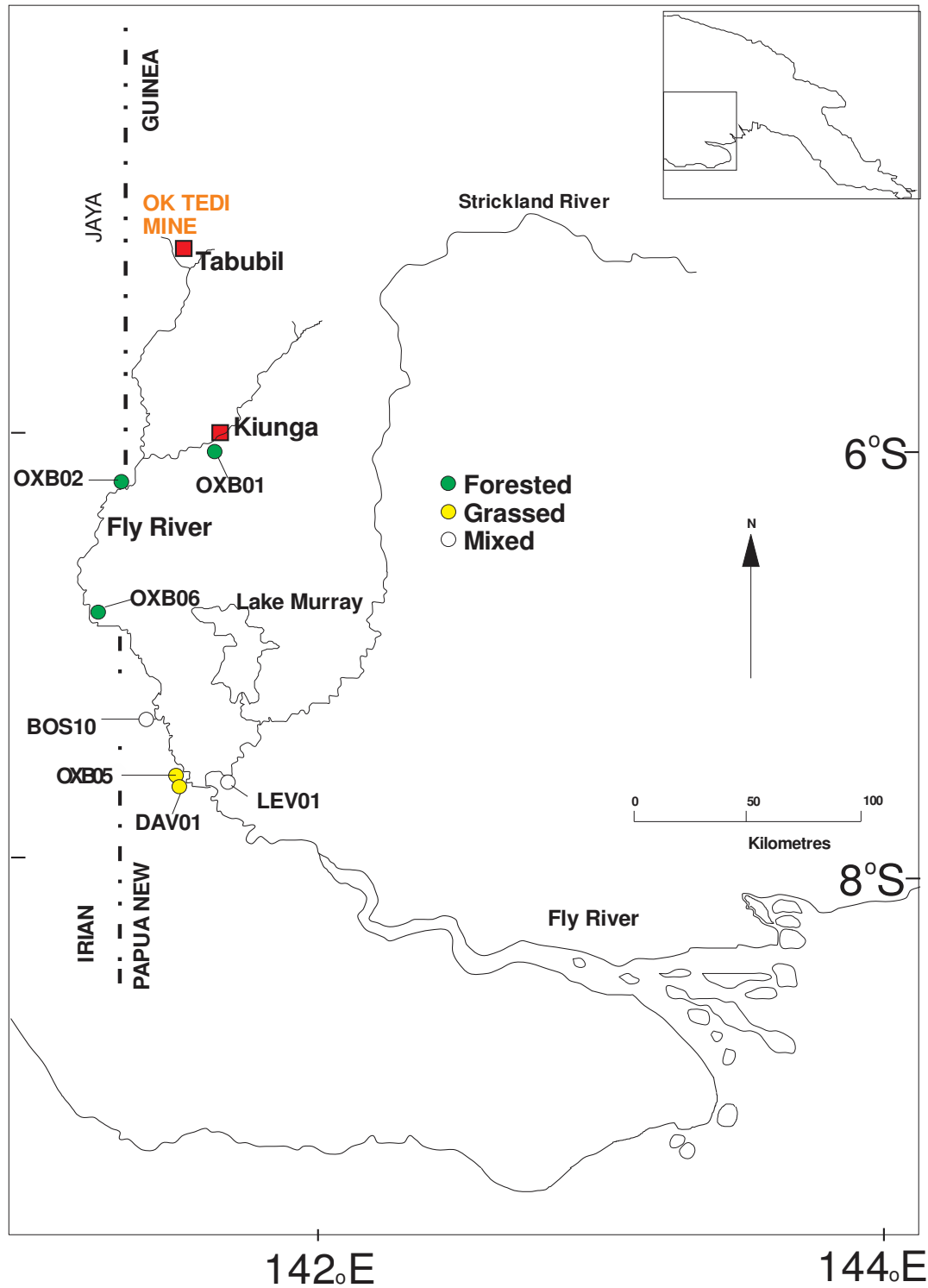
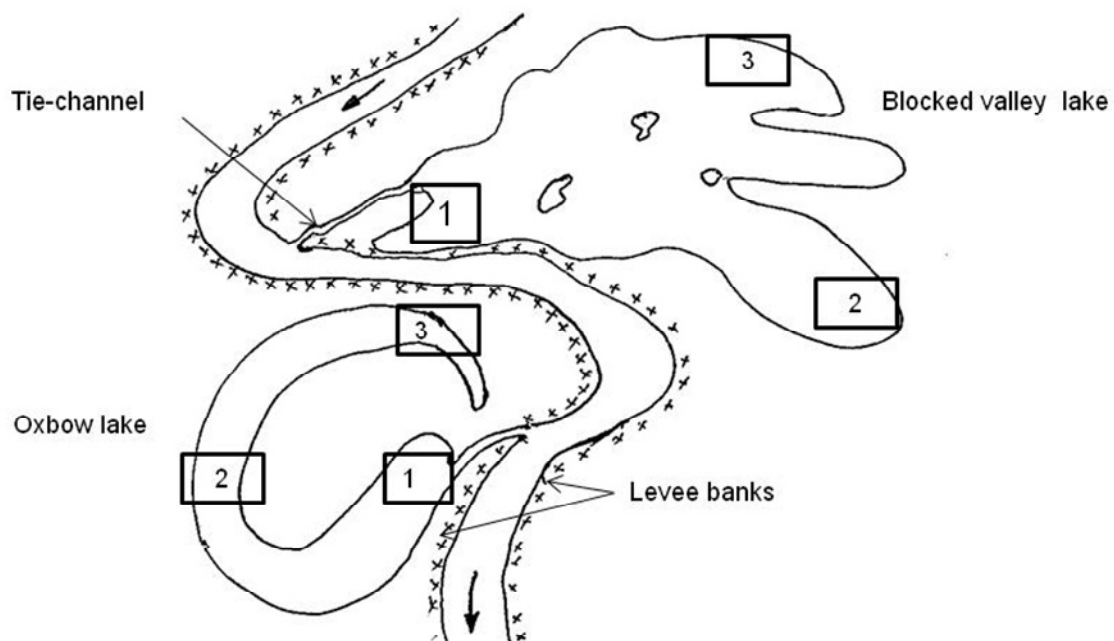


Figure 2.3 Locations of forested, grassed, and mixed vegetation lakes sampled in this study

The oxbow at Levee (LEV01) is on the Strickland River, approximately 10 km upstream of the Fly–Strickland river junction (the Everill Junction). This lake is characterised by very clear waters and is surrounded by a mixed forested/grassland type of vegetation as the forested reaches of the Strickland to the north transit into the grassland vegetation of the Fly floodplain to the south. It is connected to the Strickland via a 70 m tie-channel. This lake is not subjected to mine impacts and is regarded as a reference system.

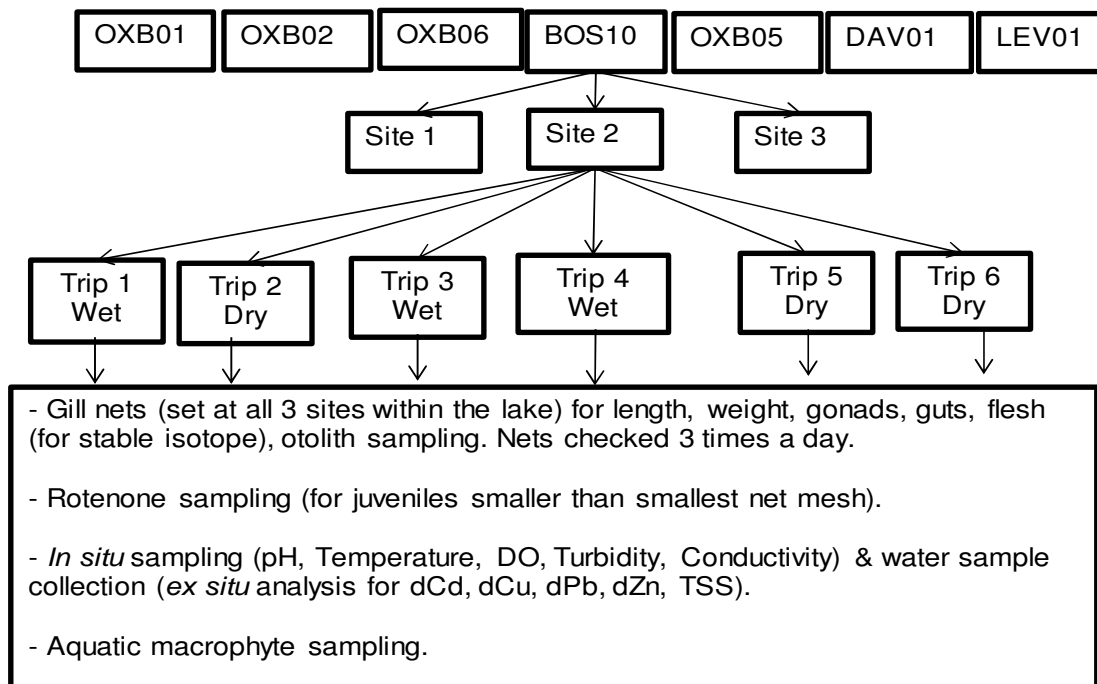
Three study sites (1, 2 & 3) were selected in each water body based on: a) their accessibility and accessibility to available data by OTML; b) their having representation from forested, transition regions (i.e. between grassed and forested reaches of the system), and grassed floodplain sites; or c) their being un-impacted lakes in the system. (Figure 2.4). For the OBLs the study sites were selected at each end of the oxbow and the middle stretch. Each study site within a lake (location) was approximately 200 m in length and 100-200 m wide, depending on the width and type of water body. The BVLs are broader so the average width of the OBLs (200 m) was used as the study width for each site within the water body.



**Figure 2.4 Locations of sampling sites 1, 2 & 3 within oxbow & blocked valley lakes. Each site in an oxbow is approximately 100 m long by the width (approximate average of 200 m) of the lake at that site. Blocked valley sites are 100 m along the shore by 200 m width (perpendicular to shoreline)**

### 2.3.2 Habitat sampling

Six sampling trips were undertaken in 2008 and 2009. At each of the seven ORWBs, three sites were sampled during the wet (3 trips) and dry (3 trips) seasons (Figure 2.5). Sampling was undertaken at each site within each ORWB to estimate vegetation cover, conduct *in situ* measurements of water quality, collect water samples for total suspended solids (TSS) and dissolved metals (Cd, Cu, Pb and Zn) for *ex situ* analysis as presented in Table 1.2. The metals were sampled because of their occurrence in mine effluent and their historical monitoring in fish tissues and water by OTML. During the study a total of 630 (7 lakes x 3 sites x 6 trips x 5 variables) *in situ* measurements were taken and a total of 630 (7 lakes x 3 replicate sites x 6 trips x 5 variables) water samples were collected for laboratory analysis.



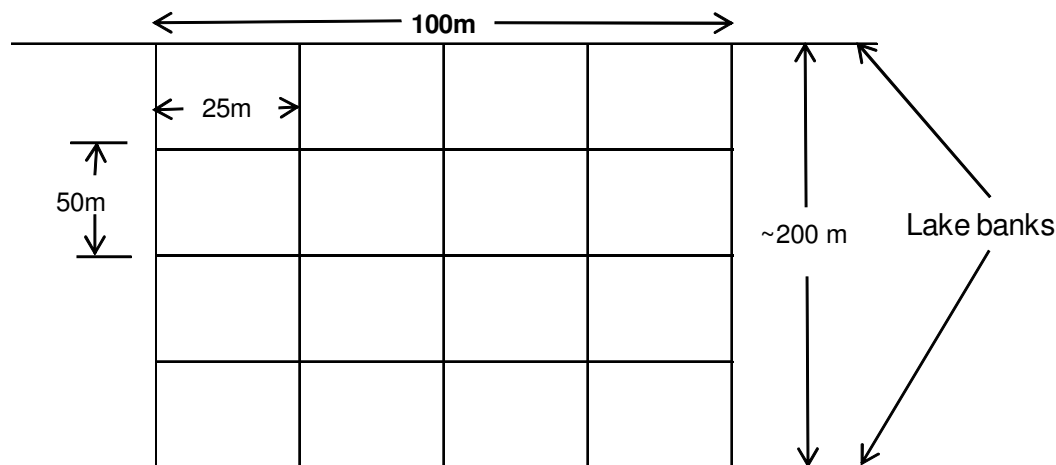
**Figure 2.5 Study design. Sampling was done for the studied variables in 3 sites within each of the 7 lakes during each of the 6 trips coinciding with the wet and dry seasons.**

#### 2.3.2.1 Vegetation cover

The average area of habitat at each of the 3 sites within a lake was 2 ha (100 m long x 200 m wide), giving an approximate study area of 6 ha per lake. Each study site was divided into a grid of 16 sampling units, the dimensions of which depended on the width of the site, but in most cases the area of each was 50 m x 25 m (Figure 2.6). A motorised dinghy was used to traverse the grid, estimating the percentage cover to the nearest 10 % for each grid in each site. A white disk attached to the end of a rope was used as

a secchi disk to give a measure of visibility in deeper and, on very rare occasions, very turbid waters. Absence of aquatic vegetation within the limit of visibility was recorded as such. The number of aquatic macrophytes species (submerged and emergent) were counted for each site. A collection of aquatic macrophytes (submerged, floating and emergent) at each lake was made for identification with the assistance of the PNG National Herbarium in Lae.

The vegetation cover estimates were averaged to give a mean percentage cover for the total study area within each lake (Figure 2.6). The same sites were sampled for fish during this study.



**Figure 2.6** Vegetation survey grid for each site. Each study site was approximately 100 m long and 200 m wide, and comprised 16 quadrats of 25 x 50 m wide.

### 2.3.2.2 Water quality

#### *In situ measurements*

Measurements of physico-chemical parameters were made in conjunction with fish sampling. All measurements were obtained with a portable Yeo-Kal Model 611 water quality analyser. Surface water temperature ( $^{\circ}\text{C}$ ), conductivity ( $\mu\text{S}/\text{cm}$ ), dissolved oxygen ( $\text{mg}/\text{L}$ ); pH and turbidity (NTU) were measured in the water column at 1 m depth at each site during each trip between 1000 and 1200 hours. Water depth was measured using a Portable DataSonde Echo sounder from a boat, held perpendicular to and touching the water surface when the readings were taken.

#### *Dissolved metals*

In association with *in situ* measurements, water samples were taken at each site using a Niskin water sampler at 1.0 m below the water surface. Samples were immediately filtered through  $0.45 \mu\text{m}$  filters to separate the particulate from the dissolved phase and stored in refrigerator before transportation to



the laboratory for additional processing and subsequent determination of dissolved metal concentrations.

In the laboratory the filtered water samples were preserved by acidification to 0.4% with a clean/double distilled nitric acid to. The acidified water samples were analysed by Inductively Coupled Plasma Mass Spectrometer (ICP-MS). Sample concentrations were measured against standard calibration curves. All quality control and quality assurance protocols were followed as per the Australian National Association of Testing Authorities (NATA) accredited procedures used by the OTML Environment Chemistry Laboratory.

#### *Total Suspended Solids*

Water samples for total suspended solids (TSS) were also collected using the Niskin water sampler. In the laboratory, the TSS concentration in water samples was determined by filtering out the particulates onto a 0.45micron filter paper of known weight. The residual material was then dried and weighed. Using the package Infolab, TSS was then calculated as per NATA procedures used by OTML Environmental Chemistry Laboratory.

### **2.3.3 Analyses**

All data were tested for normality and log-transformed if the assumption of normality was not met. Using Statistica 9 analytical package, analysis of variance was used to test for significant relationships in water quality variables between the lakes and during the different seasons.

The separation of lakes according to their physico-chemical and macrophyte assemblages was investigated using Principal Component Analysis (PCA) using the Statistica 9.0. PCA is a multivariate analysis that ordines data and is an ideal method for data with approximate linear relationships among variables. Initial analyses of data showed one group (OXB06) very different from the others essentially an outlier making interpretation of results difficult. With its exclusion the data were reanalyzed and presented.

## **2.4 Results**

### **2.4.1 General water quality**

#### **2.4.1.1 Physicochemistry**

Physico-chemical data are presented in Table 2.2. Water quality parameters were compared against the PNG Water Quality Criteria (Environment Act 2000). In the absence of specific water quality guidelines for freshwater ecosystems in Papua New Guinea, physico-chemical data were compared to ANZECC/ARMCANZ (2000) guidelines for the protection of aquatic systems in tropical Australia. A

summary of the general water quality parameters and dissolved metal concentrations are presented in Table 2.2.

### *pH*

The pH in the lakes ranged from 6.1 at BOS10 to 8.57 at OXB06 during the dry season. The lowest wet season pH was also recorded at BOS10 (6.5) and the highest wet season pH was 8.1 at OXB01. The interaction between season and lakes was because of differences in pH recordings during the different seasons at OXB01 and DAV01 (Table 2.3; Figure 2.7). The dry season at OXB01 was significantly lower compared to its dry season pH, and pH at DAV01 was significantly lower during the wet compared to its dry season pH.

### *Temperature*

Generally, lake temperatures during the wet season were higher than during the dry season (Figure 2.7). Lake Daviambu (DAV01) showed the highest mean temperatures (30.3 °C) during the wet seasons over the study period. The two top-most oxbow lakes (OXB01 and OXB02) showed the lowest mean temperature recordings (28.2 °C). Seasonal differences within lakes (Table 2.3, Figure 2.7) were only seen at OXB06, OXB05, and DAV01.

### *Dissolved Oxygen*

The PNG Water Quality Criteria for aquatic life protection states that in freshwater systems the dissolved oxygen levels must not be less than 6.0 mg/L (PNG Environment Act 2000). Dissolved oxygen (DO) in the study lakes ranged from 2.0 mg/L at DAV01 to 8.1 mg/L at OXB01. Seasonal differences seem to be present in DO levels at OXB05 and LEV01 where concentrations during the dry season are higher than during the wet. The other lakes showed no seasonal differences in DO concentrations (Table 2.3; Figure 2.7).

### *Conductivity, turbidity, and total suspended solids*

Conductivity was low at all sites and well within ANZECC/ARMCANZ guidelines whilst the highest TSS and therefore turbidity readings were recorded at OXB06. Mean turbidity recorded during the study period ranged from 2.68 NTU at BOS10 during the wet season to 40.31 at OXB06 during the dry which is the reason for significant relationship in turbidity between sites. Most measurements were below the PNG Water Quality Criteria (Environment Act 2000) which is 25 NTU. TSS concentrations ranged from 1 mg/L at OXB01 to 40 mg/L at OXB06. Lakes outside the mine impacted reach showed lowest concentrations of TSS (Table 2.3; Figure 2.7). No seasonal variations were noted in TSS at all sites.

**Table 2.1 General characteristics of the study lakes**

Site	Site Code	Latitude/Longitude Altitude:	Habitat & Water Body type	Length (km)	Width (km)	Depth (m)	Substrate	Dominant aquatic vegetation
Ulawas oxbow	OXB01	Lat: 6° 9' 53.70"S Lon: 141° 16' 44.53"E Alt: 24 m	Forested Oxbow (U)	4	Max: 0.25 Min:0.2	Max: 10 Min:6	Fine silt & clay ; heavy leaf litter from forest trees	No submerged vegetation found, only semi aquatic pandanus palms.
Erekta Oxbow	OXB02	Lat: 6° 16' 40.01"S Lon: 141° 2' 21.10"E Alt: 19 m	Forested Oxbow (I)	1	Max: 0.25 Min: 0.07	Max: 6 Min: 4	Fine silt & decaying organic matter	Some Hydrocharitaceae, and water lilies (Nymphaeaceae and Menyanthaceae) Particularly prominent were the semi aquatic pandanus palms. Littoral zones have developed from deltas formed from mine sediments.
Kwem Oxbow	OXB06	Lat: 6° 42' 6.68"S Lon: 140° 52' 21.96"E Alt: 14 m	Forested Oxbow – currently under heavy forest die back (I)	3.8	Max: 0.42 Min: 0.18	Max: >10 Min: 6	Fine silt, dark silty clay & decaying organic matter	Floating grasses (Poaceae) and water lilies (Nymphaeaceae and Menyanthaceae) found at the shallower ends of the oxbow. Encroaching floodplain grass vegetation.
Bosset Lagoon	BOS10	Lat: 7° 13' 24.72"S Lon: 141° 6' 31.18"E Alt: 7 m	Mixed forested/grassed Blocked Valley Lake (I)	7	Max: 5.1 Min: 1.3	Max: 7 Min: 4	Fine silt, dark silty clay and decaying organic matter	Dominated by the families Cyperaceae , Poaceae, and the water lily families of Nymphaeaceae and Menyanthaceae. Extensive littoral zones
Lake Pangua	OXB05	Lat: 7° 32' 2.35"S Lon: 141° 17' 19.31"E Alt: 5 m	Grassed Oxbow (I)	5	Max: 0.62 Min: 0.30	Max: >20 Min: 9	Fine silt, dark silty clay and decaying organic matter	Dominated by the families Cyperaceae , Poaceae and the water lily families( Nymphaeaceae & Menyanthaceae ) Hydrocharitaceae. Littoral zones at shallower ends of oxbow
Lake Daviambu	DAV01	Lat: 7° 35' 15.79"S Lon: 141° 17' 13.62"E Alt: 5 m	Grassed Blocked Valley Lake (I)	7.6	Max: 4.22 Min: 0.92	Max: 7 Min: 3	Fine silt, dark silty clay and decaying organic matter,	Submerged aquatic macrophytes (mainly Hydrocharitaceae); floating grass & emergent aquatic vegetation of the same families as in BOS10. Extensive littoral zones due to shallow depth
Levame oxbow	LEV01	Lat: 7° 32' 12.03"S Lon: 141° 26' 48.60"E Alt: 9 m	Mixed Forested/Grassed Oxbow (U)	5	Max: 0.41 Min: 0.28	Max: >20 Min: 9	Fine silt, dark silty clay and decaying organic matter	Only at the ends of the oxbow, where there is a bit of littoral zone. Macrophytes (mainly Hydrocharitaceae); floating grass and water lilies.

Note: I = mine impacted ; U= unimpacted

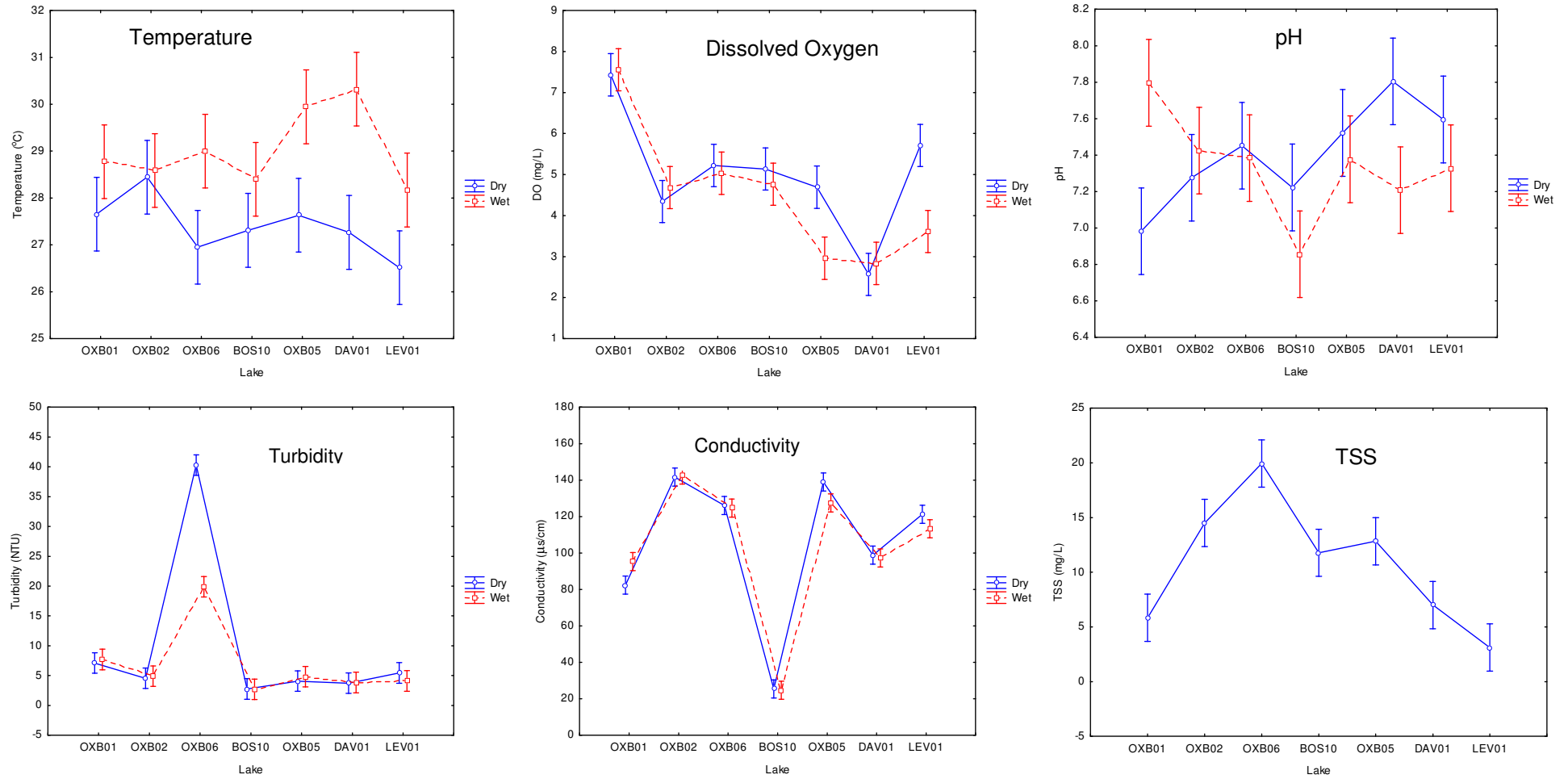
**Table 2.2 General physicochemistry of study lakes showing the ranges, the mean and standard error of these parameters.**

Habitat type	Lake	Statistic	pH		Total suspended solids (mg/L)		Temperature (°C)		Dissolved Oxygen (mg/L)		Conductivity (ms/cm)		Turbidity (NTU)		Dissolved Cu (mg/L)		Dissolved Zn (mg/L)		Dissolved Pb (mg/L)		Dissolved Cd (mg/L)	
			Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry
Forested oxbow	OXB01	Min	7.5	6.5	4	4	27.98	26.2	6.5	6.5	87	78	5.6	6.2	0.5	0.4	0.23	0.09	0.11	0.11	0.02	0.02
		Max	8.1	7.23	7	11	29.4	28.8	8.1	8	100	89	11	8	1	0.72	0.6	0.71	0.16	0.32	0.07	0.08
		<b>Mean</b>	<b>7.80</b>	<b>6.98</b>	<b>5.44</b>	<b>6.22</b>	<b>28.78</b>	<b>27.66</b>	<b>7.56</b>	<b>7.43</b>	<b>95.33</b>	<b>82.33</b>	<b>7.70</b>	<b>7.10</b>	<b>0.82</b>	<b>0.61</b>	<b>0.45</b>	<b>0.31</b>	<b>0.14</b>	<b>0.21</b>	<b>0.04</b>	<b>0.03</b>
		SE	0.08	0.08	0.38	0.78	0.19	0.31	0.22	0.18	2.09	1.69	0.84	0.25	0.06	0.04	0.05	0.09	0.00	0.02	0.01	0.01
	OXB02	Min	6.98	6.8	10	10	26.89	26.5	3.4	3	135	138	3.1	3.9	4.2	4.1	0.2	0.1	0.1	0.12	0.02	0.01
		Max	7.6	7.89	21	18	31	31.1	6.1	5.9	150	148	6.2	5.3	11	12	2.3	6	0.4	0.7	0.02	0.02
		<b>Mean</b>	<b>7.42</b>	<b>7.28</b>	<b>14.89</b>	<b>14.11</b>	<b>28.59</b>	<b>28.44</b>	<b>4.68</b>	<b>4.34</b>	<b>142.78</b>	<b>141.67</b>	<b>4.89</b>	<b>4.55</b>	<b>7.37</b>	<b>7.10</b>	<b>1.20</b>	<b>1.92</b>	<b>0.21</b>	<b>0.34</b>	<b>0.02</b>	<b>0.01</b>
		SE	0.06	0.11	1.31	1.07	0.50	0.55	0.30	0.35	1.85	1.28	0.32	0.17	0.80	0.78	0.33	0.79	0.04	0.08	0.00	0.00
	OXB06	N	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9	9
		Min	6.96	6.9	10	10	28.7	25.65	4.2	4	123	123	13	33.1	13.1	12.12	3	4.1	0.61	0.69	0.02	0.01
		Max	8	8.57	34	40	29.5	28.6	6.1	6.1	127	130	27.4	50.2	20.3	20.1	10.3	7.23	0.9	1	0.05	0.03
		<b>Mean</b>	<b>7.38</b>	<b>7.45</b>	<b>19.56</b>	<b>20.33</b>	<b>29.00</b>	<b>26.95</b>	<b>5.03</b>	<b>5.22</b>	<b>124.67</b>	<b>126.00</b>	<b>19.89</b>	<b>40.31</b>	<b>16.61</b>	<b>15.25</b>	<b>5.96</b>	<b>6.38</b>	<b>0.77</b>	<b>0.80</b>	<b>0.03</b>	<b>0.02</b>
SE	0.14	0.17	2.26	3.26	0.13	0.43	0.23	0.27	0.47	0.69	1.71	2.31	0.71	0.92	0.98	0.33	0.03	0.03	0.00	0.00		
Grassed oxbow	OXB05	Min	7	7.1	3	7	29.12	26.7	2.45	3.22	119	132	3.5	3.5	2.9	3.5	1.4	0.11	0.11	0.12	0.02	0.02
		Max	7.8	8.3	20	24	31	29	3.6	6.1	138.2	143.2	5.2	4.9	6.2	5.34	4.5	5.2	0.3	0.3	0.02	0.02
		<b>Mean</b>	<b>7.38</b>	<b>7.52</b>	<b>12.44</b>	<b>13.22</b>	<b>29.95</b>	<b>27.63</b>	<b>2.96</b>	<b>4.69</b>	<b>127.43</b>	<b>138.98</b>	<b>4.82</b>	<b>4.07</b>	<b>4.74</b>	<b>4.76</b>	<b>2.84</b>	<b>3.54</b>	<b>0.26</b>	<b>0.26</b>	<b>0.02</b>	<b>0.02</b>
		SE	0.10	0.16	2.06	1.90	0.18	0.26	0.15	0.36	2.17	1.19	0.17	0.14	0.38	0.19	0.41	0.74	0.03	0.03	0	0
Grassed blocked valley	DAV01	Min	6.7	7.6	3	5	29	26	2.1	2	83	96	2.9	3.1	2	2	0.2	0.1	0.15	0.13	0.02	0.02
		Max	7.8	8.12	12	11	31.1	29.3	3.3	3.5	115	109	5	4.1	6.7	4.5	3	1	0.68	0.4	0.02	0.02
		<b>Mean</b>	<b>7.21</b>	<b>7.80</b>	<b>6.33</b>	<b>7.67</b>	<b>30.32</b>	<b>27.27</b>	<b>2.83</b>	<b>2.57</b>	<b>97.33</b>	<b>98.89</b>	<b>3.83</b>	<b>3.73</b>	<b>4.29</b>	<b>3.69</b>	<b>1.15</b>	<b>0.69</b>	<b>0.28</b>	<b>0.28</b>	<b>0.02</b>	<b>0.02</b>
		SE	0.12	0.05	1.42	0.88	0.29	0.49	0.17	0.22	4.31	1.38	0.27	0.11	0.57	0.27	0.36	0.15	0.06	0.03	0	0
Mixed (forested & grassed) blocked valley	BOS10	Min	6.5	6.1	9	6	25	25	3.9	3.7	20	19	2.1	2.2	3	2.98	2.3	2.5	0.21	0.31	0.02	0.02
		Max	7.3	7.8	20	15	30	30	5.2	6.7	31	30	3.1	3.2	6.1	4.98	4	4.2	0.52	0.42	0.04	0.03
		<b>Mean</b>	<b>6.86</b>	<b>7.22</b>	<b>12.78</b>	<b>10.78</b>	<b>28.40</b>	<b>27.31</b>	<b>4.77</b>	<b>5.14</b>	<b>24.67</b>	<b>25.44</b>	<b>2.68</b>	<b>2.76</b>	<b>4.51</b>	<b>3.65</b>	<b>3.01</b>	<b>3.56</b>	<b>0.37</b>	<b>0.36</b>	<b>0.02</b>	<b>0.02</b>
		SE	0.09	0.16	1.32	1.02	0.58	0.55	0.14	0.36	1.45	1.12	0.12	0.11	0.01	0.00	0.20	0.22	0.04	0.01	0	0
Mixed (forested & grassed) oxbow	LEV01	Min	6.7	7.1	1	1	26.9	25	2.5	5	100	98	2.3	3	0.2	0.2	0.1	0.1	0.3	0.3	0.02	0.02
		Max	7.89	8.4	7	4	30	28	5	7	130	150	6	11	0.6	0.5	0.1	0.1	0.3	0.3	0.02	0.02
		<b>Mean</b>	<b>7.33</b>	<b>7.60</b>	<b>3.33</b>	<b>2.89</b>	<b>28.17</b>	<b>26.51</b>	<b>3.61</b>	<b>5.71</b>	<b>113.33</b>	<b>121.22</b>	<b>4.10</b>	<b>5.43</b>	<b>0.32</b>	<b>0.31</b>	<b>0.10</b>	<b>0.10</b>	<b>0.30</b>	<b>0.30</b>	<b>0.02</b>	<b>0.02</b>
		SE	0.13	0.14	0.60	0.35	0.42	0.28	0.28	0.25	3.60	5.68	0.47	0.99	0.05	0.04	0	0	0	0	0	0

The ANOVA results of the general water quality parameters show significant spatial variations at all sites for all parameters (Table 2.3). There were no seasonal differences in pH, conductivity, and TSS; whilst significant seasonal differences in temperature, dissolved oxygen and turbidity were detected. The interaction terms were significant for pH, temperature, dissolved oxygen, conductivity and turbidity, but not for TSS. Figure 2.7 illustrates the significant interaction terms. Appendix 2.3 shows the actual plot of each variable at each site during the different seasons with the critical and/or acceptable levels, as per the ANZECC 2000 water quality guidelines.

**Table 2.3 Full Factorial 2 Way ANOVA results of water quality comparisons**

Parameter	Effect	SS	df	MS	F	p
pH	Location	2.623	6	0.437	3.37	0.004
	Season	0.148	1	0.148	1.14	ns
	Location*Season	5.581	6	0.930	7.17	<0.0001
	Error	14.529	112	0.130		
Temperature	Location	30.70	6	5.10	3.61	0.003
	Season	83.90	1	83.90	59.22	<0.0001
	Location*Season	24.50	6	4.10	2.88	0.01
	Error	158.7	112	1.40		
Diss. oxygen	Location	231.12	6	38.52	63.428	<0.00001
	Season	8.61	1	8.611	14.180	0.0002
	Location*Season	26.41	6	4.402	7.249	<0.00001
	Error	68.02	112	0.607		
Conductivity	Location	169950	6	28325	499.66	<0.00001
	Season	52	1	52	0.92	ns
	Location*Season	1616	6	269	4.75	0.0002
	Error	6349	112	57		
Turbidity	Location	10220	6	1703.4	251.420	<0.00001
	Season	258.34	1	258.34	38.132	<0.00001
	Location*Season	1631.2	6	271.86	40.128	<0.00001
	Error	758.8	112	6.775		
TSS	Location	3610.38	6	601.73	28.133	<0.00001
	Season	0.130	1	0.130	0.006	ns
	Location*Season	37.65	6	6.28	0.293	ns
	Error	2395.56	112	21.39		



**Figure 2.7 Interaction plots of means ( $\pm$  95% CI) of ANOVA of general water quality parameters in & among lakes during the wet & dry seasons. There were significant relationships in Temperature ( $F(6, 112)=2.8820, p=.01193$ ), DO ( $F(6, 112)=7.2485, p=0.00000$ ), pH ( $F(6, 112)=7.1705, p<0.00001$ ), Turbidity ( $F(6, 112)=40.128, p<0.0001$ ), Conductivity ( $F(6, 112)=4.7496, p=.00024$ ) and TSS ( $F(6, 112)=7.2485, p<0.00001$ ). There were no seasonal differences detected in TSS concentrations. Sites are arranged from upstream to downstream, lines connections was to identify trends.**

## *Metals*

Concentrations of heavy metals were compared to ANZECC guidelines (2000) for the protection of aquatic systems given that metal pollution is known to adversely impact aquatic biota, including fish (Lawrence & Hemmingway, 2003; Newman & Unger, 2003). Dissolved metal levels were generally low, with the exception of copper (Cu) with mean Cu concentrations exceeding guidelines (ANZECC= 1.4 µg/L) at all exposed sites, being particularly high at OXB06. All other metal concentrations were low, or below detection limits at all sites as previously reported by WRM (2006). Appendix 2.4 shows the mean concentrations of dissolved metals at each ORWB during the wet and dry seasons over the study period.

The minimum dissolved Cu concentrations recorded during this study were from LEV01 on the Strickland for both the dry and wet seasons at 0.2 µg/L, and the maximum was from OXB06 during both the wet and dry season (20.3 and 20.1 µg/L respectively). Generally the dissolved Cu concentrations recorded during the study period at all the lakes downstream of the mine (OXB02, OXB06, BOS10, OXB05, DAV01) were significantly higher than lakes located away from the mine impacted areas (Figure 2.8). The mean Cu concentrations in all mine-impacted lakes, except for BOS10, exceeded the ANZECC/ARMCANZ guidelines (2000) for the protection of 95% of species in aquatic systems, where OXB02 was 7.23, OXB06 was 15.93, OXB05 measured 4.75, and DAV01 recorded 3.99 µg/L. Other metals' (Cd, Pb, and Zn) concentrations were all below the ranges of concern and so are not discussed further.

### **2.4.1.2 Correlation of water quality variables**

Significant regressions and correlations between water quality parameters in the study lakes were detected (Table 2.5).

#### *Temperature, pH, and dissolved oxygen*

After the removal of temperature outliers, OXB01 and LEV01 (because OXB01 is higher in elevation and closer to the cooler mountains than the rest, and LEV01 is on the Strickland river floodplain), from the dataset, a significant negative regression ( $p = 0.0115$ ) was observed between pH and temperature (Table 2.4). There was a very weak correlation between pH and temperature ( $r = -0.2728$ ). This relationship indicates that in the cooler months (dry season) there is an increase in pH, and when the temperature increases during the wet season (summer) there is a drop in pH. Temperature also had a highly significant negative relationship with dissolved oxygen ( $r = -0.467$ ,  $p < 0.00001$ ), indicating that with higher temperatures, less DO is available in the water column. These relationships are commonly known in aquatic systems where increased temperatures result in lower oxygen solubility and therefore an increase in acidity.

### *Total suspended solids and metals*

A highly significant relationship was detected between TSS and dissolved Cu ( $r = 0.8251$ ,  $p < 0.0001$ ). Highly significant regressions, and positive correlations were also seen in TSS and dissolved Pb ( $r = 0.5675$ ,  $p < 0.0001$ ), and TSS and dissolved Zn ( $r = 0.7434$ ;  $p < 0.0001$ ) (Table 2.4).

There were also significant relationships between dissolved Pb and dissolved Cu ( $r = 0.8544$ ;  $p < 0.0001$ ), dissolved Cu and dissolved Zn ( $r = 0.7668$ ;  $p < 0.0001$ ), and dissolved Zn and dissolved Pb ( $r = 0.7924$ ;  $p < 0.0001$ ). These results indicate that there is a strong link between increased TSS and increases in metal concentrations and conductivity in the mine-impacted lakes.

**Table 2.4 Significant regressions between water quality parameters and dissolved metals in the study lakes.**

Variables		Statistics					
X	Y	Slope	r	df	F	p	
Log(Temperature +1)	Log (pH+1)	-0.227	-0.273	1,83	6.674	0.012	
Temperature	Log(DO+1)	-0.032	-0.467	1,109	30.466	<0.00001	
TSS	Conductivity	1.961	0.576	1,101	50.162	<0.00001	
Log(TSS+1)	Log(DO +1)	0.237	0.481	1,76	36.644	<0.00002	
Log(TSS+1)	Log(dCu+1)	1.105	0.825	1,117	308.221	<0.00001	
TSS	dPb	0.017	0.568	1,116	55.11	<0.00001	
Log(TSS+1)	Log(dZn+1)	0.799	0.743	1,110	135.918	<0.00001	
dCu	dPb	0.035	0.854	1,105	2372.348	<0.00001	
Log(dZn+1)	dCu	13.175	0.767	1,104	148.453	<0.00001	
dZn	dPb	0.069	0.792	1,111	187.254	<0.00001	

### **2.4.2 Macrophyte species composition and cover**

A total of 45 aquatic macrophyte species were identified during the study (Appendix 2.2). The species richness at the lakes in descending order was DAV01 (40), OXB05 (38), BOS10 (32), LEV01 (30), OXB02 (13), OXB06 (9), and OXB01 (8). The grassed floodplain lakes had the greatest number of species compared to the forested oxbows.

Table 2.5 shows that of the 45 species of aquatic macrophytes identified (Appendix 2.2), more than 70% were found in each of the grassed floodplain lakes. The blocked valley lakes, DAV01 and BOS10, had the highest percentage vegetation cover, with 65.42 % and 53.54 % respectively. Table 2.5 also shows that site 2 in all oxbow lakes had the least vegetation cover due to the absence of littoral zones in the middle section of these lakes. The ANOVA results presented in Table 2.6 illustrate this relationship where there is significant difference in vegetation cover and species counts between grassed floodplain lakes and the forested lakes. The results of Tukey's tests show clearly that the grassed floodplain lakes had higher species diversity and vegetation cover than the forested lakes



**Table 2.5 Summary of lake study area, species composition and vegetation cover in the Fly River lakes.**

Lake	Lake Study area (ha)	Site	Site study area (ha)	Vegetated area (m <sup>2</sup> )	Site species count	Total species	Site percent cover	Mean percent cover of study area
OXB01	6.00	1	2	1125	4	8	6	4.59
		2	2	375	3		1.88	
		3	2	1250	6		6.25	
OXB02	2.40	1	0.8	1500	6	13	18.75	17.09
		2	1	812.5	8		8.13	
		3	0.8	1950	10		24.38	
OXB06	5.00	1	1	1750	5	9	17.5	14.88
		2	2.5	1000	2		4	
		3	1.5	3468.75	8		23.13	
BOS10	4.40	1	1.5	9750	18	32	65	53.54
		2	2	7250	23		36.25	
		3	0.9	5343.75	16		59.38	
OXB05	6.50	1	2	4500	19	38	22.5	21.04
		2	2.5	2968.75	13		11.88	
		3	2	5,750	25		28.75	
DAV01	6.00	1	2	13500	23	40	67.5	65.42
		2	2	13625	25		68.13	
		3	2	12125	32		60.63	
LEV01	6.00	1	2	2375	10	30	11.88	13.02
		2	2.5	1875	9		9.38	
		3	1.5	3562.5	21		17.81	

**Table 2.6 One-way ANOVA of species counts and percent vegetation cover at all study lakes. Significant relationships were investigated further by Tukey's test. Tukey's test results are arranged in descending order of means with lines joining lakes that are not significantly different.**

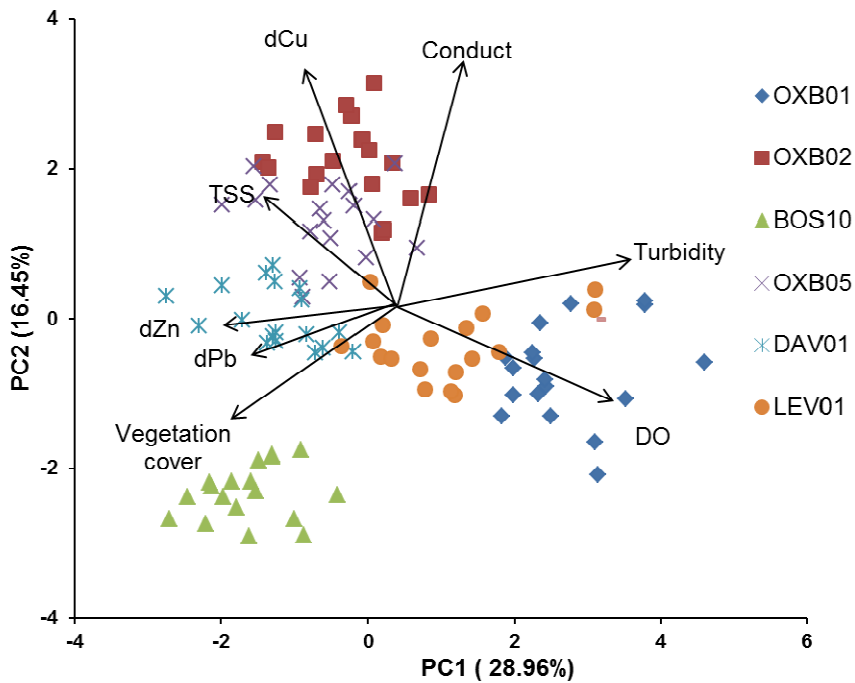
Comparison	Effect	SS	df	MS	F	p	Tukey's HSD Test
Species count	Lake	1.312	6	0.219	15.318	<0.0001	DAV01 BOS10 OXB05 LEV01 OXB02 OXB06 OXB01
	Error	0.199	14	0.014			
Percent Cover	Lake	2.590	6	0.432	9.835	<0.001	DAV01 BOS10 OXB05 OXB02 LEV01 OXB06 OXB01
	Error	0.614	14	0.044			

### **2.4.3 Ordination of lakes by biophysical parameters**

The initial PCA for all sites showed a highly skewed result with the strong influence of OXB06. Subsequent PCA on all the vegetation and water quality data reduced the physico-chemical dataset to three principal components, with over 45% of the variation being explained by PC1 and PC2 (Table 2.8; Figure 2.9b).

With OXB06 included in the analysis 48.42 % of the variation was explained by PC1 and PC2; the main responsible parameters were copper, turbidity, lead, zinc, and total suspended solids (Table 2.8). With the exclusion of OXB06, 45.30% was explained by PC1 and PC2; with the responsible parameters on axis 1 being turbidity, oxygen, vegetation cover, and zinc. Distribution along axis 2 was influenced by conductivity and copper (Table 2.8). Figure 2.9a shows that OXB06 is an outlier and once it is removed from the analysis the separation of the lakes according to the environmental parameters was clearer.

The results show that water quality at lakes closer to the mine (OXB02, OXB06) and along the mine-impacted reaches (OXB05, DAV01) were strongly influenced by mine-derived effluents. These lakes had greater concentrations of dissolved metals, conductivity, and TSS in comparison to lakes outside the mine-impacted region such as OXB01, which is upstream of the mine waste source, and LEV01 on the Strickland River floodplain. BOS10 was separated more by its vegetation cover than its other water quality parameters, even though it is located along the mine-impacted reach, right after OXB06. The other BVL, DAV01, was separated both by the influence of water quality and also its vegetation cover. The non-mine-impacted lakes, OXB01 and LEV01, were separated by their higher turbidity and DO respectively.



**Figure 2.8** Principal Component Analysis of habitat parameters showing the separation of the lakes (without OXB06) and key environmental vectors. Percent of variance explained by each PC is indicated in the axis labels.

**Table 2.7 Correlations for habitat parameters corresponding to different axes of PCA.**

Significant r values are shown in bold for analysis with and without OXB06, which was heavily influenced by mine-derived effluents.

Parameter	With OXB06			Without OXB06		
	Percent Variance explained					
	Axis 1 32.67%	Axis 2 15.75%	Axis 3 13.82%	Axis 1 28.96%	Axis 2 16.45%	Axis 3 12.65%
Vegetation cover	0.182	<b>0.851</b>	0.081	<b>-0.682</b>	-0.421	0.257
pH	-0.055	-0.277	-0.411	0.298	0.357	0.363
TSS	<b>-0.753</b>	0.100	0.049	<b>-0.543</b>	0.433	<b>-0.586</b>
Temperature	0.064	0.264	-0.279	-0.165	0.213	0.196
Dissolved oxgen	0.060	<b>-0.697</b>	<b>0.571</b>	<b>0.702</b>	-0.193	-0.467
Conductivity	-0.392	-0.450	<b>-0.709</b>	0.195	<b>0.861</b>	0.250
Turbidity	<b>-0.831</b>	-0.154	0.194	<b>0.733</b>	0.269	-0.174
Dissolved copper	<b>-0.918</b>	0.003	-0.176	<b>-0.702</b>	<b>0.499</b>	-0.184
Dissolved zinc	<b>-0.777</b>	0.255	0.261	<b>-0.620</b>	0.031	<b>-0.516</b>
Dissolved lead	<b>-0.826</b>	0.187	0.172	-0.464	-0.286	0.251
Dissolved cadmium	0.097	-0.198	<b>0.518</b>	0.386	-0.299	-0.371

## 2.5 Discussion

### 2.5.1 Water quality

#### 2.5.1.1 Physicochemistry

Water quality alteration in the world's freshwater and marine environments due to changes in land use, industrial activities and climate are of notable concern. Water quality can be affected by changes in nutrient composition, changes in the natural physicochemical properties of the water body, or through the invasion of pristine environments by pollutants associated with anthropogenic activities (e.g. Kadlec, 2003; Thomas *et al*, 2003; Kantawanichkul & Neamkam, 2003)

When fish and other organisms experience environmental disturbances which lie outside the normal range the effects can be dramatic, and long-term exposure to less severe disturbances may result in severe consequences to individuals in populations (Jobling, 1995). Nutrients, metals, hydrocarbons etc. may have various effects on the individuals as well as having long term implications for the

ecosystem. Any changes in an aquatic environment are usually dictated by the prevailing physicochemistry of the medium, consequently affecting the flora and fauna (Meyer, 2002; Yaru & Buckney, 2003).

Analyses of current data show that there are significant location differences in water quality variables at all lakes in this study. Some of these physicochemical parameters, such as pH, temperature, and DO are most probably associated more with the location of the lakes, whether upstream and in the forested reach or downstream in the grassed floodplain, or due to seasonal effects. Other variables such as turbidity, conductivity, and TSS, are undoubtedly more clearly influenced by mine impacts on the receiving environment.

This impact on the water quality downstream of the mine has been reported previously (Storey *et al*, 2009a; Bolton *et al*, 2009; Stauber & Apte, 2009), especially as a result mine effluents being directly discharged into the Fly River system. These impacts are manifested in the elevated concentrations of copper, total suspended solids, zinc, and conductivity in some lakes as shown in this study.

Significant seasonal variation in mean dissolved oxygen and temperature, where cooler temperatures were consistently recorded during the dry season (southern winter) months with correspondingly slightly higher dissolved oxygen recorded during those months, is the first record of this nature from the Fly River lakes. The lakes with the lowest temperatures are also located at elevations which are higher than the floodplain lakes. The Ulawas lake (OXB01), at 24 m a.s.l, is closer to the cooler Fly waters from the headwaters and the Levame oxbow (LEV01) receives water from the Strickland, which is a cooler faster flowing river than the Fly. This is interesting because of the importance that temperature and DO play in the biology of aquatic species (Lloyd, 1992, Wetzel, 2001), and may explain the community structure and species presence or absence in different habitats.

Even though the mean pH in the ORWBs remains between 6.8 and 7.8, it is clear that there are different trends during the different seasons. The patterns show that the pH is higher in the forested lakes during the wet season and decreases with distance downstream into the grassland lakes. This trend is then seen to be reversed during the dry season. It is important to note that even small shifts in pH in an environment which is under stress from pollutants could mean significant consequences as a result of impacts of pollutants such as heavy metals when the environment turns acidic as reported elsewhere in literature (Jobling, 1995; Schindler, 1996; Meyer *et al*, 1999). This situation could be exacerbated further during prolonged dry seasons under the climate change scenario where the water bodies increase in acidity even more as a result of higher temperatures and therefore lower DO.

Correlations of different variables reveal that there is a significant negative relationship between temperature and DO, and temperature and pH recordings. The explanation for this is that during the wet season, when the temperatures are usually higher in the grassed floodplain lakes, increased microbial decomposition of organic matter causes DO and pH to drop (Lewis, Jr, 1987; Marotta *et al*,

2010). Temperature, water pH, alkalinity and hardness can have a significant effect on the solubility and therefore toxicity of a given pesticide, herbicide or heavy metal influence. The toxicity of most metals is reduced as pH increases, because at high pH the metal ions complex out, thus rendering them unavailable; however, in most water bodies hardness, alkalinity and pH vary together so the effects of these factors on metals tend to be interwoven (Jobling, 1995).

Changes in temperature and DO content of water have been reported to affect fish resistance to toxic chemicals or diseases (e.g. Bucke, 1993), while changes in temperature, pH, water hardness, and humic acids can affect the toxic state of some chemicals or compete with their uptake by fish (e.g. Lloyd, 1992). In addition, environmental conditions such as temperature, day length and food availability can have major effects on the kinetics of uptake, biotransformation and elimination processes of toxicants such as metals (Baykan *et al*, 2007). It is also known that temperature has strong effects on animal activity such as respiration, feeding, growth and reproduction with temperature effects such that it also controls the uptake and elimination process of xenobiotics (e.g. Baykan *et al*, 2007). Temperature increases have also been found to have a profound effect on physiological mechanisms of fish larvae, primarily through changes in the rates of enzymatic reactions (Hochachka & Somero, 2001).

Furthermore, Breitburg *et al* (1997) found that low but non-lethal DO concentrations greatly increased predation on fish larvae. They also noted that predator-prey interactions reflected variation among species in their physiological tolerance to low oxygen and the effects of low oxygen on the escape behavior of prey, as well as on swimming and feeding behaviours of predators.

Water quality parameters such as oxygen play an important role in community structure. Hypoxia can change fish assemblage composition and may lead to mortality if fish cannot escape to more oxygenated waters (Smale & Rabeni, 1995 cited in Killgore & Hoover, 2001). It has been reported, for example, that species richness, abundance, and size of fish were substantially reduced at hypoxic levels, suggesting a threshold response level of assemblage composition (Jobling, 1995; Killgore & Hoover, 2001). Hypoxic conditions have also been linked to fish development retardation and lengthening of incubation time of early egg stages of some fish (Oseid & Smith, 1971; Rosenthal & Alderdice, 1976; Giorgi & Congleton, 1984; Keckeis *et al*, 1996)

According to local Middle Fly villagers, historically, the wet season periods (especially between December and February) were usually the times of the year when massive fish kills were reported in the Fly River system. The importance of this observation is that with the predicted increases in temperature around the equatorial regions due to climate change would result further in prolonged low DO concentrations and therefore increases in acidity of these lakes. The consequence of this on the fisheries ecology of this system could be detrimental considering the fact that most of these lakes are

recipients of metal-bearing sediments which will release toxic metals, such as, Cu with lake acidification.

It is not surprising that the BVLs, BOS10 and DAV01, are the ones with long-term low mean pH recordings, as they also have the highest percentage vegetation cover, and therefore dissolved and undissolved dead organic matter. Both Bosset Lagoon (BOS10) and Lake Daviambu (DAV01) are very broad shallow lakes that are heavily vegetated. This dense cover of macrophytes is a feature of all blocked valley lakes in the Fly River system. Apte (2009) noted that the high dissolved organic matter in these lakes is probably the main reason for their slightly acidic nature. Therefore, even in the presence of high amount of organic matter, which would have acted as organic complexing agents for the metals, the lower pH of the BVLs would have been responsible for the continuous release of metals into the water column, thereby resulting in higher than normal concentrations of dissolved metals such as Cu in dissolved form. These metals concentrations could not be a result of oxidation because these lakes' beds are usually under water, and not exposed, all year round, with the exception of extreme *El Niño* induced droughts.

There is also a significant positive relationship between TSS and DO indicating that DO increases with TSS. This is an important observation because with an increase in sedimentation it is expected that DO would decrease. And, with a predicted increase in flow into this river system as a result of climate change there will be much more sediment delivered onto the floodplains and the effects of increased sedimentation and its metal loads is of concern. This may need to be investigated further because as seen previously and from OTML (unpublished data) historical data, DO is naturally lower in the grassed floodplain lakes compared to the oxbow lakes in the forested reaches and at elevation.

### **2.5.1.2 Dissolved metals**

The increased metal concentrations in the water bodies clearly reflect the mine impacts on the receiving environment. The main reason that the lake metal concentrations did not exceed levels in the channel waters is mainly due to the complexing capacity of the lakes, being high in dissolved organic matter (Apte, 2009). The risk, however, of acid rock drainage (ARD) affecting aquatic life in the ORWBs is already a reality when mine-derived sediments become exposed to air during the dry season and are oxidised, creating an acidic environment that will affect biota (Bolton *et al*, 2009; Storey *et al*, 2009a). It is known that acidic conditions can then cause leaching of complexed metals rendering them toxic to aquatic biota.

Although the levels of Zn, Pb and Cd are below levels that may be of concern to aquatic organisms, there is a clear difference in levels of Cu, Pb, and Zn between mine impacted and non-mine impacted lakes, indicating external contamination of the lakes. This, however, could also be a result of natural variations or be indicative of a mineralised region. Various reports have shown that there has been an

increase in metal content of the Fly channel waters since mining commenced in the headwaters of the Ok Tedi (Day *et al.*; 1993; Yaru *et al.*, 1999; Yaru & Buckney, 2000; Apte, 2009). A recent study by WRM (2005) showed that there is strong mine signature in the lakes downstream of the mine, by virtue of the increased concentration of metals such as Cu, Pb, Zn, Pb, Ni, and Cr. This study also indicated elevated concentration of metals in some of the lakes downstream of the mine. This is a scenario common to many rivers and water bodies around the world in which habitats downstream of point-source pollution are impacted by heavy metals (e.g. Kime, 1995; Schulz & Martins-Junior, 2001; Benejam *et al.*, 2010). This is of concern because excessive levels of metals can have a number of effects on organisms such as: mutagenic/genotoxic effects, overcoming xenobiotic metabolic pathways, and affecting glycolysis, Krebs cycle, oxidative phosphorylation, protein amino acid metabolism as well as carbohydrate and lipid metabolism (Codina *et al.*, 1994; Jannasch *et al.*, 1999; Almeida *et al.*, 2001; Sanchez-Galen *et al.*, 2001; Vernhet *et al.*, 2001; Pouls, 2002; Ohe *et al.*, 2004; Styrdom *et al.*, 2006).

The results of this study also clearly established a strong correlation between TSS and dCu, dZn, and dPb downstream of the mine. It is obvious that with increased mine-derived sedimentation in the floodplains there will also be an increase in metal loads, as shown in previous studies (Yaru & Buckney, 2000; Day *et al.*, 2008; Day *et al.*, 2009). Higgins (1990) postulated that the floodplain lakes of large river systems downstream of pollutant sources act as sinks for suspended sediments with their heavy metal loads. This increase in sedimentation, and the associated metal loads, also causes increased conductivity in mine-impacted lakes, where it is also strongly correlated with TSS, as recorded in this study.

The metal of concern in these water bodies is Cu and particularly its bioavailable fraction as there has been an increase in labile Cu concentrations in the Middle Fly region since 1996 (Stauber *et al.*, 2009). There may be a number of reasons for this increase, such as more reactive copper being put through the mill, resuspension of Cu in sediments by dredging in the Lower Ok Tedi, a decrease in organic matter as a result of increased vegetation dieback in the Middle Fly region which then results in less Cu removed from the water column by organic matter through complexation (Stauber *et al.* 2009). Stauber *et al.* (2009) also reported a corresponding increased inhibition of algal growth, which may result in 50-80% of freshwater species in this region facing some form of chronic toxicity issues. Rogers *et al.* (2005) suggested that if there are any further increases in bioavailable Cu in the Fly River, the majority of the freshwater species would be subjected to chronic toxicity symptoms.

### **2.5.1.3 Macrophytes**

Most of the aquatic macrophyte species recorded in this study were collected from BVLs, lakes with more extensive littoral zones than oxbow lakes. These results indicated that the grassed floodplain lakes had more habitat heterogeneity, and are possibly more important habitats for resident lacustrine



than migrant species compared to the forested oxbow lakes. The results of PCA ordination shows that mine impacts and vegetation cover were crucial biophysical parameters that separated the lakes. Lakes closer to the mining outfall are much more prone to mining effects, and have been subjected to increased habitat degradation especially from sedimentation (Storey *et al*, 2009), and also face the prospect of ARD. While numerous reports have been produced on the effects of mine effluents on fish communities in the Fly River, the few that have dealt with metal-plant interactions in the Fly River ORWBs (e.g. Yaru *et al*, 1999; Yaru & Buckney, 2003) have also shown that aquatic macrophytes have been active in the filtering and/or taking up metals from sediments and/or solution. The consequence of this to higher organisms such as fish has not been examined. On the other hand, work on toxicity of metals to algae has been ongoing since 1996 (Stauber *et al*, 2009).

Aquatic macrophytes are important determinants in the structuring of littoral communities (Goulding 1980, Ara'ujo-Lima *et al*, 1986, Rosas & Odum 1987, Savino & Stein 1989, Bettoli *et al*, 1992). The presence of macrophytes, undoubtedly, provides more habitats for many organisms as well as fish (Johnson, 1993, Savino & Stein 1982, Werner *et al*, 1983) and also contributes to primary production in these lakes. It is in these shallow vegetated areas that adult fish as well as juveniles of many species seek refuge as well as food for their various life stages (Dewey *et al*. 1997, Weaver *et al*. 1997; Hatzenbeler *et al*. 2000; Vono & Barbosa, 2001; Pelicice *et al*, 2008; Neiff *et al*, 2009). Of these floodplain lakes, those with extended littoral zones are usually the most productive, such as the blocked valley lakes, in comparison to the deeper oxbow lakes with limited or no littoral zones (Vono & Barbosa, 2001).

Not only are the macrophytes useful as habitats and carbon sources for aquatic species, they have been clearly identified as important to the protection of water bodies by trapping and removing domestic and industrial wastes (Reddy & De Busk, 1985; De Busk *et al*., 1989; Tripathi & Shukla, 1991).

Work by Yaru *et al* (1999) showed that aquatic macrophytes are important in the filtering of metals transported in the water column, and in the uptake of metals deposited in sediments in the Fly River floodplain. In doing that, the aquatic macrophytes, and riparian vegetation act as important barriers against the entry of pollutants into the floodplain water bodies. However, with the predicted increase in overbank flooding in the Middle Fly over the next 40 years, as the sediment front and its metal load progresses further downstream, the threat to all forms of aquatic species becomes greater as terrestrial and aquatic vegetation composition changes with prolonged flooding (Pickup, 2009).

## 2.6 Conclusion

This study has shown that water bodies along this impacted river corridor have been affected by mining discharges with lakes nearest the mine having higher levels of metals and TSS than sites further away from the mining outfalls. Although the changes and pressures in large tropical river systems like the Fly River are likely to be similar to those in temperate systems, little study has been undertaken to verify this or to determine how the outcomes of these pressures might be altered in a hot and humid tropical setting, or in the face of diverse tropical fauna and complex food webs.

Consequently, the implications of various impact scenarios on key ecosystem components are unknown. In particular, ecological threats to key species, such as phytodetrivores, which feed at the base of the food chain, are poorly understood. Similarly, their responses to environmental changes are unknown, even though this is likely to offer insights into how aquatic food webs will be impacted and so allow robust predictions and the development of responsible management plans. It is therefore vital for ecological and socioeconomic reasons to understand in some detail the background and ecology of species like *N. papuensis*.

In understanding the un-impacted as well as disturbed habitats of this impacted corridor proper fisheries and habitat management plans could be put in place to conserve aquatic flora and fauna of this region for use by current and future generations. With an increase in forest and riparian vegetation die-back, these impacts have the potential to spread further into the other floodplain habitats with continuous riverbed aggradation and under certain climate change predictions for this region.

## **Chapter 3 Distribution and abundance of *Nematalosa papuensis* populations of floodplain water bodies**

### **3.1 Introduction**

The need to understand fish community and population structure and distribution is crucially important in the face of challenges faced by confined ecosystems such as freshwater lakes and rivers, which are subjected to increasing environmental and fishing pressure for both commercial and food security purposes. This is particularly true in large tropical river/lake systems, where there are high human population densities, and traditional fisheries form a critical staple means for local peoples' food security. These are also the regions where there are rapid natural resource extraction and development resulting in greater stress on the aquatic systems (e.g. Adeyamo, 2003).

In many of these large river systems the major issue has been that of habitat destruction or alteration (Roni *et al.*, 2005), which culminates in a variety of impacts on fish at the cellular, organ, individual, population and eventually community levels of organisation (Lloyd, 1992; Lawrence & Hemmingway, 2003; Newman & Unger, 2003), and can impact their spatial and temporal dynamics.

Few fish spend their entire life in the same habitat (Welcomme & Cowx, 1998) but usually move around to find microhabitat conditions that are suitable for each life stage or time of year. Consequently, fish spatial and temporal dynamics are in part functions of habitat structure and conditions (e.g. Bishop & Harland, 1982; Schlosser, 1982; Harris & Kangas, 1988; Bishop & Forbes, 1991; Welcomme & Cowx, 1998; Buffagni *et al.*, 2000). A number of useful models have been proposed to explain community dynamics and structures according to a range of spatial and temporal factors (Jackson *et al.*, 2001), leading to a set of theories explaining community variations in rivers and streams, and patterns of productivity along lateral and longitudinal gradients (Vannote *et al.*, 1980; Schlosser, 1982; Junk *et al.*, 1989; Thorp & DeLong, 1994). Fish distribution and movements has been described in a four dimensional framework – i.e. longitudinal, lateral, vertical, and temporal (seasonal) – illustrating the dynamic spatial and temporal nature of fish movements and habitat requirements in freshwater systems. (e.g., Welcomme & Cowx, 1998; Silvano *et al.*, 2000; Rolls & Wilson, 2010).

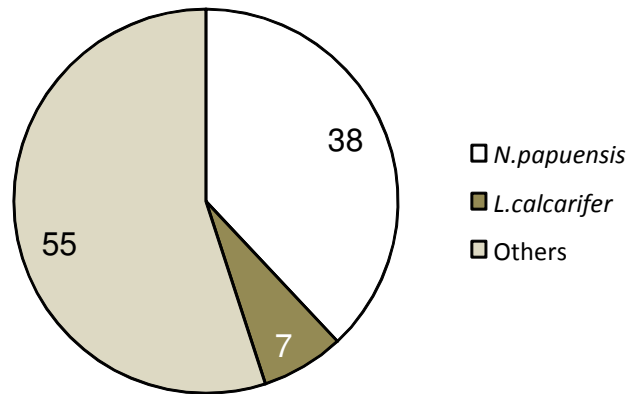
The Fly River floodplain/wetland system is the largest system within the New Guinea-Australia and South Pacific zoogeographic regions, totaling over 4.5 million hectares (Ellison, 2009). The floodplain water bodies of this system produce much higher fish catches and have greater species diversity than riverine areas (Storey, 2005; Hortle & Storey, 2006). A characteristic of the fish fauna of these lakes is that there are significant inter-annual variations in the fish catches in the shallower BVLs and GFL in comparison to the deeper OBL, especially during extreme climatic events such as the *El Niños*

(Swales *et al.*, 1999; Storey *et al.*, 2009b). To date, few detailed investigations have been undertaken into how temporal changes during wet and dry seasons affect the dynamics of fish populations in these lakes. Oxbow lakes are the exception: being deeper, they retain water through drought periods and therefore provide a drought refuge for fish (Smith & Bakowa, 1993; Swales *et al.*, 1999).

Differences in fish communities in the different floodplain habitats probably reflect different habitat structures and the tendency for the GFL and shallower BLVs to dry out in times of extreme droughts (Swales *et al.*, 1999) as water recedes and the habitat contracts (Mol *et al.*, 2000). Despite these extreme seasonal changes floodplain water bodies comprise a complex suite of habitats that support a diverse assemblage of freshwater fish, with 66 species from 33 families recorded (Swales *et al.*, 1999). As in the main river channel, fish diversity on the floodplain is dominated by catfish of the families Ariidae (11 species) and Plotosidae (7 species). Numerically, however *N. papuensis* is the single most dominant species in the Fly River system, both in the channel and ORWB catches (OTML, 1995).

Despite its numerical dominance in the Fly River fisheries ecology (Figure 3.1), *N. papuensis* has not received any detailed attention apart from studies investigating fisheries options (e.g., Wilson, 1992; Hortle & Storey, 2006). These studies suggest the potential for a fishery to be developed with an estimated yield of 5,000-18,000 tonnes per annum with little impact on the sustainability of this species. These studies are based on estimates of abundance. However, there is no documentation of the dynamics of this species related to seasonality or habitat differences (i.e. lake types), factors which need to be understood before fisheries viability can be fully assessed. This issue is of concern for many developing countries of the tropics where the ecology of target species has not been fully defined through thorough understanding prior to the establishment of fisheries.

Such studies in other regions have provided useful information for understanding fisheries ecology and management. For example, an understanding of the distribution and abundance of fish can be a reflection of habitat complexity and food availability (Grenouillet *et al.*, 2002); an understanding of the community composition, abundance and trophic structure would provide insight into fisheries ecology and fish community structure (Balanga-Robles *et al.*, 2002); and spatio-temporal based sampling of populations reveals seasonal assemblage structures (Fialho *et al.*, 2007). Fish assemblages in large river systems over space and time as a result of environmental influences can also be gleaned by such studies (Araújo *et al.*, 2009). Additionally, fishery management tools can be established if catch effort data could be established from this kind of study as has been done for clupeids in large African lakes (e.g. Coulter, 1970; Marshall, 1988), to Asia (De Graaf, 2003; De Silva, S.S & Funge-Smith, 2005)



**Figure 3.1 Main species biomass composition of Fly River ORWBs showing the contribution of *N. papuensis* (OTML data, pers. comm).**

As a hyper-abundant, key component of the Fly River fauna, *N. papuensis* is a useful model species for the development of clear understanding of the likely impacts of this complex of threats on phytodetritivorous fish in tropical freshwater systems, and on the food webs in which they are keystone components. An understanding of such a species and its habitat requirements has implications for the management and conservation of fisheries in similar tropical systems in New Guinea and elsewhere. The intent of this chapter is to investigate the key drivers for the spatial and temporal dynamics and distribution of this species over the last 20 years, and during the 2 years when this study was undertaken, aiming to enhance the management of fisheries locally and in similar systems elsewhere.

This project involved catching of fish and was undertaken under Ethics Approval Number A1353 granted by the JCU Animal Ethics Committee.

## 3.2 Methods

### 3.2.1 Sampling design

Six sampling trips were conducted between 2008 and 2009 to coincide with the wet and dry seasons as explained in the study design in Chapter 2, section 2.3.2. Fish were sampled at each replicate site of the 7 lakes during each trip. Monsoonal rains on the floodplains fall from December to May and the dry period is from June to November. The sites data were used as replicates for each lake and the trip data were used as replicates for each season in a two-way factorial design.

Three sets of gill nets of six mesh sizes (Table 3.1) were set concurrently for 12 hours (0600 – 1800 hours) at three random sites within the lakes (see Chapter 2, Figure 2.4). The net dimensions used in this study correspond with those used by Ok Tedi Mining Limited (OTML) for over 20 years in the Fly River system (Smith & Bakowa 1993), to allow comparisons between current data and the historical dataset. The second reason for using these net sizes is that historically these nets contribute over 97% of all *N. papuensis* catches from the off-river water bodies (OTML unpublished data). The nets were checked every 4 hours to give three repeat samples for biomass and number of fish caught during each trip at each site. Because these 3 repeat samples were not independent, they were combined to give a sample representative of the 12-hour period.

Data from a pilot study demonstrated that the catch per unit effort for the current design did not differ significantly from the historical catches, which were based on a single net set of similar mesh sizes being left in one site within the lake for 24 hours. The method used in this study was aimed at reducing predation of fish in the net and to reduce the possibility of reduction in catches if the nets became choked with captured fish. Initially 4 trips per year were planned; however, due to the loss of 2 sets of samples because of freezer failure, data from only 3 trips in 2008 and 3 trips in 2009 are reported in this study. The approach therefore follows a two-way factorial design with lakes and seasons as factors and the catch data from sites within lakes and the three trips each season used as replicates for each lake and season respectively.

**Table 3.1 Dimensions of monofilament nylon nets used in this study**

Mesh (mm)	Net Dimensions		
	Length (m)	Depth (m)	Area (m <sup>2</sup> )
25	40	2.3	92
38	40	1.7	68
50	45	2.1	95
63	40	2.8	112
75	45	3.2	144
88	45	3.5	158

### 3.2.2 Fish sampling

The nets were set following OTML protocols, from the smallest to the largest mesh size, allowing a space of between 40-50 m between nets. The end of the nets were tied to poles or grasses on the lake edge and anchored by concrete anchors perpendicular to the banks. Three nets each were set on either side of the banks (for an oxbow). Because the BVLs are much wider, the second set of three nets was usually located in open water by tying onto bamboo poles set into the lake bed approximately 200 m

(equivalent to average width of oxbow lakes) from the shore. The nets were checked and cleared every 4 hours in order to maintain specimen freshness.

To be consistent with historical sampling methods, the fish toxicant rotenone was used to sample for juveniles at all locations during each trip. Since 1989, rotenone has been used by OTML following a standardised protocol at each monitoring location. For each sample, 500 g of powdered rotenone was mixed with approximately 10 L of water and a small quantity of detergent. The mixture was then applied to a small, sheltered bay (~ 0.1 ha) and affected fish were collected by means of dip-nets from a small boat for one hour after the first affected fish were taken. The same location was fished on each occasion. The method can only be regarded as qualitative, because the water depth and dense vegetation precluded the containment of a known area.

Historical data from OTML for the same net mesh dimensions, from 1988-2002, was used to determine the *N. papuensis* distribution trends at OXB06, BOS10, OXB05 and DAV01 with respect to influence of climatic patterns and possible mine impacts on the environment.

### **3.2.3 Processing of fish**

All net-caught fish were handled fresh from capture. Fish lengths (length to caudal fork - LCF) were measured to the nearest 1 mm using a fish measuring board. Each fish was then blotted dry and weighed on a calibrated digital electronic balance accurate to 0.1 g. Net mesh size used in the capture of each fish was recorded along with its length and weight. Juveniles were preserved in ice and returned to the laboratory for measurements of length (to 1 mm) and weight (to 0.01 g). Surplus fish were given to nearby villagers. Fish that were classified as juveniles were those less than 70 mm in length (length at caudal fork). The 70 mm size cut-off point was the length at which some fish matured sexually (i.e. reached late stage 2 and early stage 3) (Chapter 6).

### **3.2.4 Data analysis**

#### **3.2.4.1 Short- term variations**

The 2008-2009 data were used to examine the current patterns of population variation within and between water bodies. Since there was a very strong correlation between biomass and abundance (see below) only the biomass data were used in the analysis. The catch data were log transformed ( $\log_e(\text{biomass}+1)$ ) to normalise data before being subjected to further analysis. A 2-Way ANOVA was used to determine differences between lakes (habitats) and seasons for the pooled fish catches in each lake. Significant interactions were presented by water body type (oxbows and blocked valley lakes) to give a clear picture of habitat patterns. A 2-Way ANOVA was also used to compare juvenile catches between lakes and seasons. Lakes where no juveniles (OXB01), or only few (only 13 in OXB02), were collected were treated as outliers and excluded from analyses.

### 3.2.4.2 Assessment of historical data

Historical fish catch data were transformed in a similar manner to the 2008-2009 dataset before analysis. This data-set was used to assess spatial and temporal population fluctuations and probable changes in response to seasonal and/or climate influenced hydrological patterns (wet/dry, *El Niño/La Niña*), and mine-related impacts in the ORWBs.

One-way ANOVA was used to determine between-year differences. Long-term trends were investigated by simple linear regression analysis for sites with the available historical data after the exclusion of *El Niño* fish catch data as outliers, where some data points are zero while catches at other times for lakes such as OBLs are extremely high. A lagged phase plot analysis of the long-term data, including the catch data from *El Niño* years, against the *El Niño* Southern Oscillation Index (ENSO) was used to investigate if these climatic conditions might have caused long term changes in fish populations.

Unfortunately there were few records of water levels for the floodplain sites available for correlation analysis.

## 3.3 Results

### 3.3.1 Short-term patterns

During the study 17 361 adult and 2164 juvenile *N. papuensis* were collected, with a combined total weight of 1008.1 kg. A summary of *N. papuensis* catches during the different seasons at the different lakes is presented in Figure 3.3. The juvenile data are summarised in Figure 3.4.

#### 3.3.1.1 Patterns in different habitats

##### *Total catches*

Fish catch data collected during this study showed low mean biomass in forested oxbows as opposed to grassed floodplain lakes (Figure 3.3, Appendix 3.1). From the forested lakes, mean biomass yield during the study period was 2.846 kg from OXB01, 34.959 kg from OXB02, and 35.362 kg from OXB06. The floodplain lakes, in comparison, yielded 96.072 kg at BOS10, 76.683 kg at OXB05, 39.615 kg at DAV01, and 47.694 kg at LEV01 on the Strickland. A correlation analysis of  $\text{Log}_e(\text{biomass}+1)$  and  $\text{Log}_e(\text{numbers}+1)$  detected a very strong correlation ( $p < 0.001$ ;  $r^2 = 0.96$ ), indicating that for this species either catch per unit effort numbers or biomass could be used to describe the population dynamics.

A comparison of these data sets showed a significant interaction between lakes and seasons ( $p < 0.0001$ ; Table 3.2). Tukey's HSD post hoc tests showed that OXB01 was clearly different from the rest. OXB02 did not differ from wet season catches at DAV01, and catches in BOS10 during wet and



dry periods, and OXB05 catches during the dry seasons did not differ from each other and were higher than all the other sites during the different seasons. Most lakes showed a general trend of higher mean catches during the dry than wet season; however, these seasonal variations were only significant for catches in OXB05 and DAV01 (Table 3.2; Figure 3.2).

The data also indicate a general trend of increasing catches away from forested lakes into the grassed floodplain lakes (Figure 3.2).

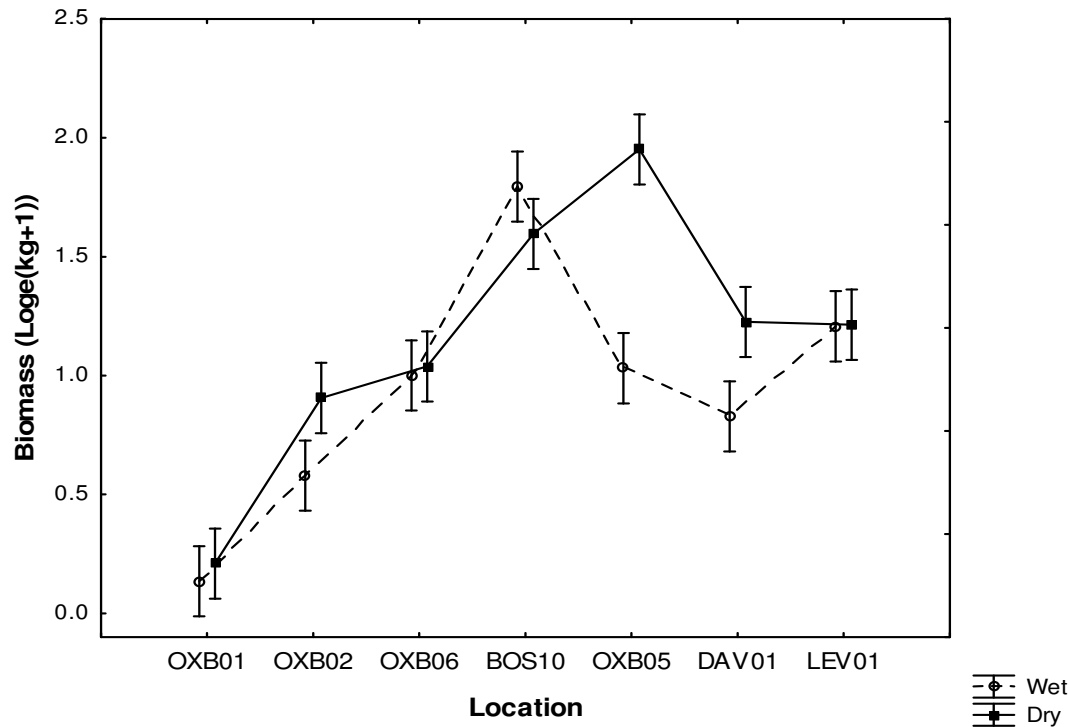


Figure 3.2 Mean weight ( $\pm 95\%CI$ ) of catches of *N. papuensis* in each lake during the wet and dry seasons.

Table 3.2 Results of 2-Way ANOVA for weight of *N. papuensis* caught during the study.

Effect	SS	df	MS	F	p
Location	81.131	6	12.072	65.972	<0.00001
Trip	8.572	5	1.294	7.073	<0.00001
Location*Trip	31.956	30	1.379	7.533	<0.0000
Error	30.442	84	0.183		

### Tukey's HSD Test

OXB01 W OXB01 D OXB02 W DAV01 W OXB02 D OXB06 W OXB05 W OXB06 D LEV01 W LEV01 D DAV01 D BOS10 D BOS10 W OXB05 D

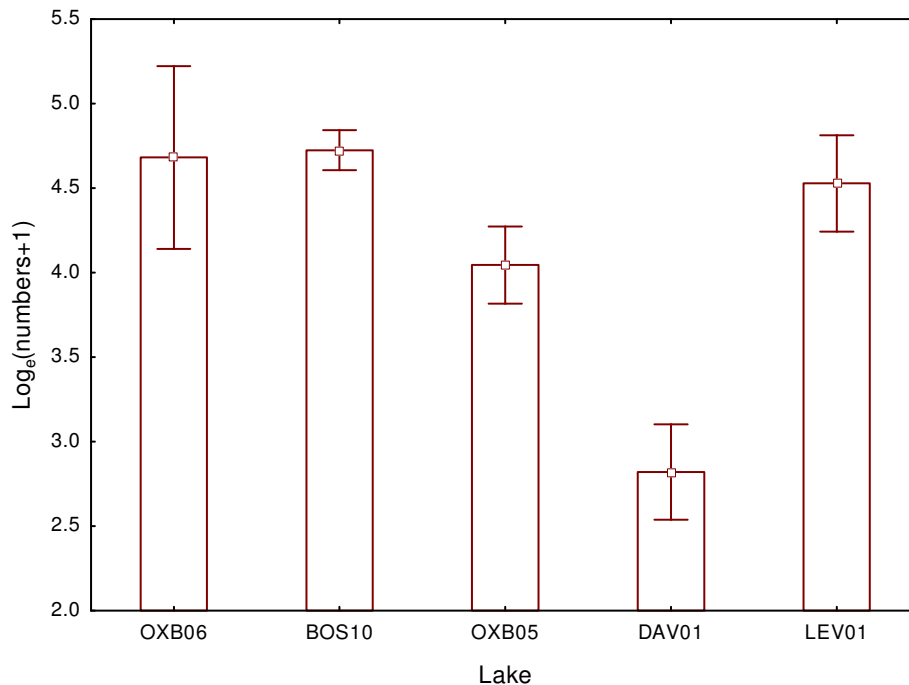
### 3.3.1.2 Juvenile catches

A 2-way ANOVA showed a highly significant difference between lakes for juvenile catches ( $p < 0.00001$ ; Table 3.3; Figure 3.4). There was, however, no seasonal difference in number of juveniles caught ( $p > 0.05$ ; Table 3.5).

Figure 3.4 shows that the greatest mean numbers of juveniles sampled during the study period were from BOS10, followed by OXB06, LEV01, OXB05, and DAV01. This significant location difference is due to very low juvenile catches at DAV01

**Table 3.3 Results of 2-Way ANOVA on juveniles sampled from 5 of the 7 lakes. OXB01 and OXB02 (two forested lakes) were excluded as outliers in this analysis**

Effect	SS	df	MS	F	p
Location	14.2603	4	3.5651	10.562	0.000370
Season	0.8513	1	0.8513	2.522	0.134586
Location*Season	2.3451	4	0.5863	1.737	0.197828
Error	4.7255	14	0.3375		



**Figure 3.3 Mean number of juveniles ( $\pm 1$  SE) caught at each lake. Lakes are arranged in order of increasing distance downstream from the mine. Two forested lakes (OXB01 and OXB02) were excluded from the analysis as outliers.**

### 3.3.2 Long-term dynamics

A total of 48742 individuals, including juveniles, weighing a total of 4749.3 kg, were collected from OXB06, BOS10, OXB05 and DAV01 between 1988 and 2002 (Appendix 3.2). Of these, 3343.1 kg were caught during the dry seasons and 1406.2 kg were caught during the wet seasons. The dataset was incomplete for two of the sites (OXB01 and OXB02) because sampling at these locations was discontinued due to heavy fishing pressure from locals (OXB01) and heavy sedimentation from mine impacts (OXB02). No juveniles were recorded from these two locations. Sampling at OXB06 only commenced in 1993. The study lake on the Strickland River (LEV01) was a new site selected for this study and therefore did not have historical data. Sampling effort at these lakes was not consistent, mainly due to logistical issues and landowner disruptions, which resulted in very low numbers of samples at some lakes, and high numbers at others (see Appendix 3.2), so any comparisons of habitats and seasons is not possible

#### 3.3.1.3 Long-term changes

One-way ANOVA showed that there was a significant difference in catches at OXB01, with lower mean catches in 2008 and 2009 than in 1990 (Table 3.4). The 2008 *N. papuensis* catches at this site were significantly lower than those of 1990, whilst the catches in 1990 did not differ from those of 2009. Catches at BOS10 were very high in 1992 and 1993 compared to catches in the dry (1995) and *El Niño* years of 1997/98.

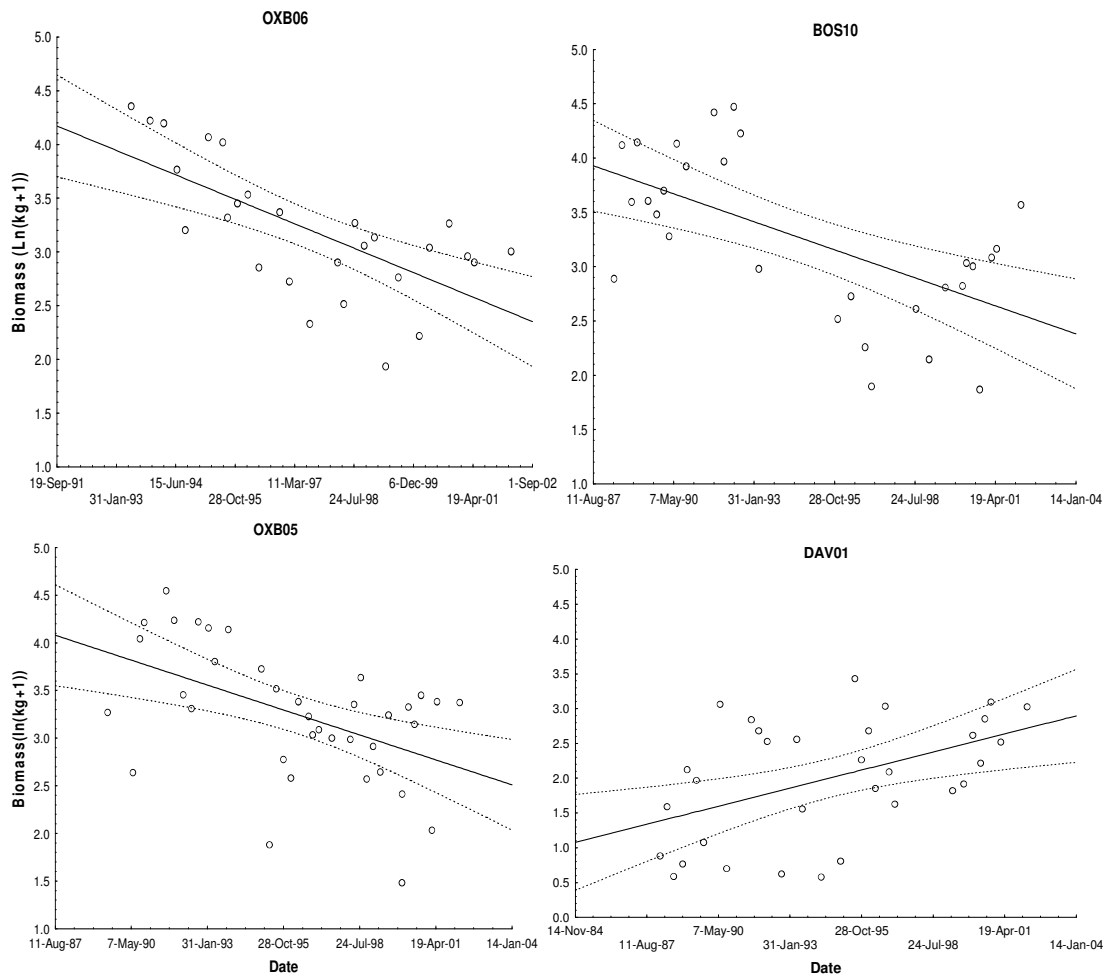
Long-term catch trend analysis show that catches in three of the lakes downstream of the mine were on a downward trend, two of them (OXB06 and BOS10) significantly so ( $p=0.0015$  and  $0.0004$  respectively) (Table 3.5; Figure 3.5).

**Table 3.4 Summary of one-way ANOVA of each lake comparing historical biomass catch data with the 2008/09 catch data. Tukey's HSD post hoc test was used to determine variation between years. Lines connect years not significantly different from each other and are arranged in descending order of the mean. ns = non-significant.**

Lake	Earliest dataset	Latest data	No. of years	N	dF	F	p	Tukey's HSD Test
OXB01	1990	2009	3	9	2,7	5.393	0.030	_____ 1990    2009    2008
OXB02	1990	2009	3	9	2,7	0.230	0.890	ns
OXB06	1993	2009	12	34	11,22	2.050	0.073	ns
BOS10	1988	2009	16	39	15,23	5.600	<0.0001	_____ _____ 91 92 93 09 08 90 89 02 88 01 99 96 00 98 95 97
OXB05	1989	2009	16	30	12,17	1.218	0.319	ns
DAV01	1988	2009	17	45	16,28	2.176	0.035	ns

**Table 3.5 Summary of Regression analyses of long-term trends in catches at historical lakes.**

Lake	N	df	F	p	R <sup>2</sup>	Slope
OXB06	28	1,26	12.648	0.0015	0.327	-ve
BOS10	33	1,31	15.569	0.0004	0.334	-ve
OXB05	38	1,36	3.503	0.069	0.089	-ve
DAV01	32	1,30	9.063	0.005	0.232	+ve



**Figure 3.4** Long-term biomass (kg) trends with 95% confidence limits in *N. papuensis* catches at the historical sites over time. Figures were plotted after exclusion of data influenced by *El Niño*

Results shown in Table 3.5 and Figure 3.5 indicate long-term declines in fish catches at OXB06, BOS10, and OXB05, whereas catches at DAV01 were positive. The Lake at Kwem (OXB06) and Bosset Lagoon (BOS10) are closer to the mining outfall and are located within the region of severe forested and riparian vegetation die-back (Marshall, 2007). It should be noted that during the *El Niños* of 1993/1994 and 1997/1998, BOS10 was inaccessible because the tie-channel from the river into the water body dried up due to low water levels. The apparent negative trend in Lake Pangua (OXB05) is not significant ( $p=0.069$ ). Lake Daviambu (DAV01) is the only site that showed a positive growth in *N. papuensis* biomass, after the droughts of 1993/94 and 1997/98.

### 3.3.1.4 Climate impacts

A lagged phase plot of the long-term data against the *El Niño* Southern Oscillation Index (ENSO) indicated that *El Niño* periods in the Fly River system had an influence on *N. papuensis* catches in OXB06 only. The data indicated strong negative changes in biomass at OXB06 over time, while catches in BOS10 and OXB05 were quite variable over time but generally in the downward direction. Fish catches in DAV01 were fairly consistent over time. The data also showed that some of the highest catches in OXB06, BOS10, and OXB05 were recorded when the ENSO was negative, that is during the drier periods on the Fly River floodplain (Figure 3.6).

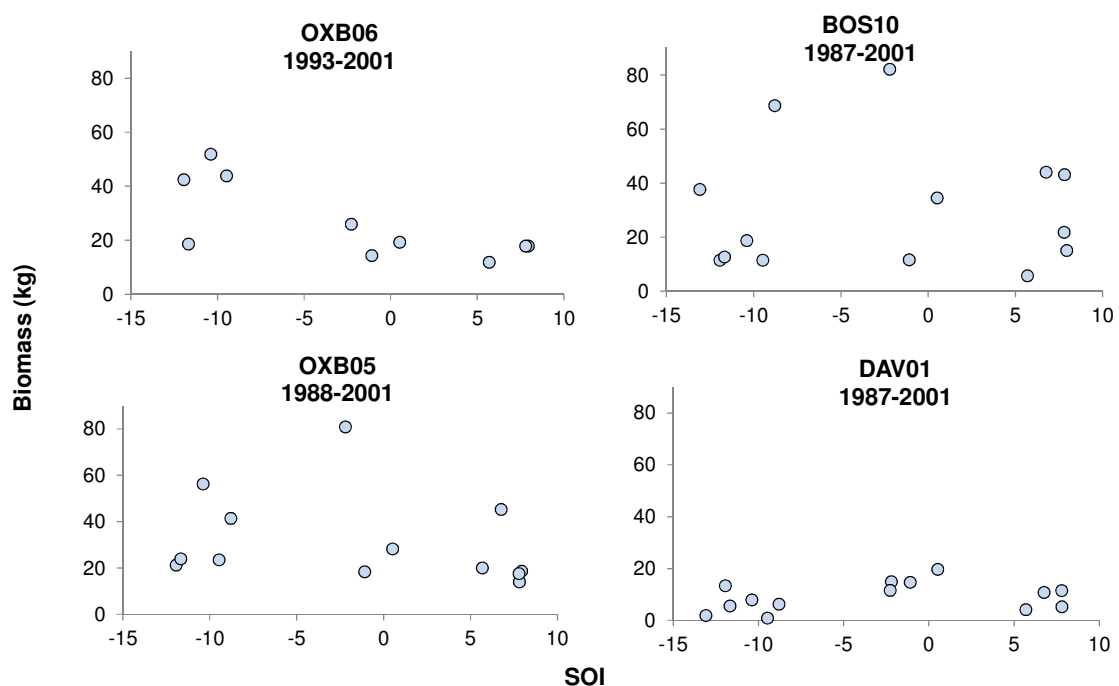


Figure 3.5 Plot of fish catch changes in *N. papuensis* biomass in the 4 lakes with the longest historical data against the Southern Oscillation Index (SOI)

## **3.4 Discussion**

### **3.4.1 Short-term spatial and temporal patterns**

#### **3.4.1.1 Habitat**

##### *Adult catches*

It is known that habitat types play a significant role in determining fish abundance, biomass, diversity, and population structure: for example, fish fauna composition and structure in different lakes of large floodplain wetland systems like the Amazon are known to differ according to habitat differences (Saint-Paul *et al.*, 2000; Petry *et al.*, 2003; Martelo *et al.*, 2008). Floodplain lake habitats typically yield larger catches than other habitats, indicating their greater level of productivity (Humphries *et al.*, 1999; Silvano *et al.*, 2000; Tockner & Standford, 2002; Storey *et al.*, 2009b). Consistent with these observations, the general trend seen from this study is that there is an increasing gradient in *N.papuensis* yield from the strictly forested OBLs into the grassed floodplain lakes. This difference in fish catches between the grassed floodplain lakes and forested lakes was first reported by Smith & Bakowa (1993) in the early 1990's when they reported very low catches at forested lakes inclusive of OXB01, whilst the fish catch and diversity was greater in the grassed floodplain lakes.

However, this trend may also be a result of fishing pressure because of the proximity of these (forested) lakes to large villages and a mining town. This is supported by the fact that during all study trips it was noticed that stretched nets (38-75 mm nets) were left in the water 24 hours a day by local villagers to catch fish either for their own consumption or for the town market. This factor was not otherwise taken into account in this study.

The higher average fish catches at BOS10 compared to other lakes were most probably due to the size of this water body, it being the largest lake in this study, offering more resources to cater for a larger population size. The catches at the other blocked valley lake, DAV01, seem to be more influenced by the natural physicochemistry and lack of space due to dense growth of macrophytes (Chapter 2). This lake had a very dense vegetation cover, and could have low dissolved oxygen levels (Chapter 2); given that *N. papuensis* is sensitive to low DO concentrations (e.g. Swales *et al.*, 1998). Nevertheless, catches in general showed similar trends to the observations by Storey (2005) and Hortle & Storey (2006), who noted that there has been a continuous decline in total fish biomass (of all species) at OXB06, BOS10, OXB05, and DAV01, lakes which are downstream of the mine.

Various physical attributes of water bodies such as habitat heterogeneity (Hortle & Lake, 1983; Capone & Kushlan, 1991; Sheldon & Meffe, 1995; Townsend, 1996; Willis *et al.*, 2005), riparian vegetation and input (Casatti *et al.*, 2003; Uieda & Uieda, 2001), and aquatic vegetation (Bishop & Harland, 1982; Bishop & Forbes, 1991) play a role in determining fish community structure and assemblages, whilst others such as water depth, substrate, and current velocity consistently appear to be important determinants of microhabitats used by stream fishes (Welcomme & Cowx, 1998). There is greater habitat heterogeneity in the grassed floodplain lakes compared to the forested lakes (Chapter 2) and this is possibly one of a number of reasons for the higher fish yield in the grassed floodplain habitats as opposed to the forested lakes. The results of this study also reflect the fact that habitat type and diversity play a significant role in determining species diversity and fish numbers in the large floodplain rivers of the world (e.g. Schlosser, 1982; Welcomme, 1985; Silvano *et al.*, 2000).

Water quality issues affecting lakes closest to the mining outfall would also impact the habitats of these lakes, thereby negatively affecting species diversity and yield (Storey *et al.*, 2009a). Also, forested lakes (downstream of the mine) are more influenced by mine-derived pollutants than other lakes (Chapter 2), because of their vicinity to the mining outfall, and this factor may have contributed to the trends observed in this study. The decrease in catches observed in OXB01 may also be indirectly related to mine impacts even though it is 41 km upstream of the mining outfall. Marshall (2006) showed that this site (OXB01) is also under stress because of backwater effects (the Fly River being pushed upstream due to sediment build-up at the confluence of the Fly and Ok Tedi). The effects of mining activities on fish have been documented in numerous publications. For example, impacts of a single mine upstream of a river system can seriously alter fish habitats fisheries downstream (Daniel *et al.*, 2015). Changes in nutrient composition, the natural physicochemical properties of the water body, or the invasion of pristine environments by pollutants result in ecosystem changes (e.g., Kadlec, 2003; Thomas *et al.*, 2003; Kantawanichkul & Neamkam, 2003). Fish and other organisms exposed to these habitat changes outside the normal range over long term may result in severe consequences to individuals in populations (Jobling, 1995).

#### *Juvenile catches*

Results showed that there was a significant difference in juvenile catches between the lakes. The strictly forested lakes, OXB02 and OXB01 had zero or very poor recruitment, respectively, while the mixed and grassed vegetated floodplain lakes generally had high numbers, again reflecting the patterns associated with habitat differences as already discussed. In this study *N. papuensis*



juveniles were found to be associated more with lakes that have shallower littoral, vegetated zones than the deeper unvegetated areas. Similarly, *N. erebi* juveniles in Australia are more abundant in shallow backwaters of billabongs (Puckridge & Walker, 1990; Bishop *et al.*, 2001), as are juvenile fish in other floodplain systems (Bonetto, 1986; Agostinho *et al.*, 2000; Neiff *et al.*, 2009).

The forested oxbow at Kwem (OXB06) had very high juvenile abundance possibly due to its location in the transition zone, where it is connected to a very large grassed/forested blocked valley wetland to the east which is drained by the Agu and Binge Rivers – tributaries of the Fly. This wetland could be the source of adults that frequent this oxbow to spawn as they would need more open space for spawning (Smith, 1998). Alternatively, since this oxbow has had severe riparian forest die-back, macrophytes more typical of grassed floodplains, such as water lilies, have encroached upon it and therefore may have created greater habitat diversity for nursery and feeding. All the other grassed floodplain lakes, apart from DAV01, had high abundance of juveniles. The low juvenile catches at DAV01 may be due to the persistently low dissolved oxygen levels in this lake (Chapter 2), possibly due to the heavy presence of macrophytes that would have high nocturnal demand for dissolved oxygen (Swales *et al.*, 1998; Hortle & Storey, 2006). Swales *et al.* (1998) have reported fish kills in the Fly River lakes due to low dissolved oxygen with *N. papuensis* being the first to die. Nonetheless, it is clear from the findings of this study that lakes in the grassed floodplain provide better habitat conditions for spawning, nursing, and recruitment of the herring, with a greater juvenile abundance found in the grassed lakes and mixed vegetated lakes of the floodplain.

#### **3.4.1.2 Seasonal catches**

Catches were greater in the dry season than in the wet for some lakes (OXB02, OXB05, and DAV01). Plots of catches in the different habitats of forested and grassed floodplain lakes showed substantial variability in catches at most sites with no consistent patterns seen for either the wet or dry season. Long term data available at OTML for the Fly River fish fauna does not show clearly any seasonal spatial differences in fish catches because of the high river and lake levels all year round. Nevertheless, studies elsewhere have shown that fish catches are very closely related to the expansion and shrinking of habitats during the wet and dry periods. For example, in the lakes of the Upper Jurua River in Brazil, fish abundance and diversity were greatest during the dry season due to the concentration of fish into the lagoons as the habitat and foraging grounds decreased (Silvano *et al.*, 2000; Saint-Paul *et al.*, 2000). Studies in temperate systems show the universality of this trend (e.g., Humphries *et al.*, 1999; Collingsworth & Kohler,

2010). The area of floodplain inundated, due to seasonal changes in water level, has a profound influence on floodplain fisheries in other river systems around the world, as well as the Fly River. The influence of *El Niño* droughts is well documented and is a major cause of some Fly River ORWBs (Bosset Lagoon (BOS10) and Lake Daviambu (DAV01)) drying out, whilst others remain flooded at all times (Ulawas Lake (OXB01), Erehta Lake (OXB02), Lake Pangua (OXB05), Kwem Oxbow (OXB06) and the Strickland site at Levame (LEV01)) (Swales *et al.*, 2000).

In contrast to the general adult population, the juveniles of *N. papuensis* did not show any significant difference in seasonal abundance. This indicates that there is continuous successful spawning throughout the year, but it is not known how successful the recruitment of juveniles into the general population is under the current biophysical conditions of the Fly River lakes.

### **3.4.2 Long-term spatial and temporal patterns**

#### **3.4.2.1 Habitat difference**

A comparative analysis of different lakes could not be undertaken because the historical datasets had very uneven sampling records, with the number of sampling trips at all lakes varying from 9 (OXB01 and OXB02) to 45 trips (DAV01). There were a number of reasons for the cessation of sampling, or aborting of trips to these lakes, which included tie-channel blockages by debris, landowner issues, and unavailability of work boats (e.g. OTML, 1995; 1996; 1997). Despite this, drawing from historical work by Smith & Bakowa (1993), and the findings of this study, there is a strong evidence that there has always been a strong difference in fish yields due to habitat effects, with forested lakes having lower yields than grass floodplain lakes, as is the case in floodplain systems elsewhere (Humphries *et al.*, 1999; Silvano *et al.*, 2000; Tockner & Standford, 2002; Storey *et al.*, 2009b).

#### **3.4.2.2 Temporal differences**

Without long-term records of fish catches at OXB01 and OXB02, there could not be confidence in the fish catch trends observed at these lakes. However, the comparison of differences in catches through time at lakes with long-term fish catch records showed that BOS10 had highly significant yearly differences in *N. papuensis* yields, with the lowest catches during the *El Niño* years (1997/98, and 1993-5). The long-term trends at OXB06, BOS10, and OXB05 all indicate a negative trend overtime. This observation is in agreement with Storey *et al.* (2009), who reported that there has been a decline in fish catches and diversity at these lakes over time, attributable to possible mine impacts.

Fish catches at DAV01 were the only ones that showed a positive trend over time. This trend may be because any impacts from the mine (from sedimentation or dissolved metals) are mitigated by the buffering/filtering effect of its large aquatic macrophyte cover and its high dissolved organic matter content (Yaru *et al*, 1999; Apte, 2009).

### **3.4.2.3 Climate influence**

Swales *et al* (1999) reported that the extreme *El Niño* conditions of the drought years resulted in very low river and lake levels with most shallow floodplain habitats drying up and an algal bloom that suffocated the lakes and affected fish catches (Swales *et al*, 1999). The exceptions were the oxbow lakes, which, being deeper provided drought refuges (Smith & Bakowa, 1993; Swales *et al*, 1999). The impact of the intensity of these events depends on the type of water body, with the larger and shallower blocked valley lakes such as Bosset Lagoon (BOS10) and Lake Daviambu (DAV01) drying out significantly during the droughts, when catches fell to zero and took 18 - 24 months to recover (Swales *et al*, 1999). The algal blooms, as seen in the Fly River lakes during the 1994 drought, were more likely a result of the increase in nutrients from fish waste as a result of fish congregating and overcrowding in the deeper water bodies, feeding and excreting wastes that caused the blue-green algae to bloom (see e.g., Glynn, 1988; Mol *et al*, 2000).

The long recolonisation period of the re-flooded lagoons is due to floating grasses, which colonise the lagoons during the drought, still covering large areas of the lakes for extended periods, inhibiting fish migration with their respiration and decay causing low dissolved oxygen levels (Swales *et al*, 1999). In some cases, the recovery of fish populations may take many years after large fish kills (Roberts & Sainty, 1996). The oxbow lakes, on the other hand, are deeper and provide refuge for all lacustrine species during extreme weather conditions, which is the reason for the peaks in OBL catches during the drought periods. The differences in fish numbers and biomass from the different lake habitats likely reflect differences in physical habitat, and the tendency for the grassed floodplain and shallower blocked valley lakes (3–4 m deep) to dry out in times of drought.

Mol *et al* (2000) reported that droughts associated with *El Niños* negatively affected fishes of lagoons, freshwater swamps, and rainforest creeks of Suriname during the 1997/98 drought. Prolonged droughts have been known to affect many species in many ways including spawning, growth, and population dynamics, and can severely affect multiple spawners (such as *N. papuensis*) (Mathews, 1998). Glynn (1988) reported that during the *El Niño* of 1982/83 food was depleted, water temperatures were higher, nutrients were depleted, and DO also decreased due to

the oxidation of organic matter produced in the euphotic zone sulphur bacteria. Post *El Niño*, he reported that there were significant changes in species abundance resulting in trophic restructuring when there was loss of dominant species. Droughts cause a reduction in the volume of water and the availability of habitats, and alter water quality, thereby altering the biology of aquatic organisms and the ecology of the lake (Mol *et al*, 2000).

From this study there is no strong evidence to show that there has been a significant climatic influence on *N. papuensis* catches in the Fly River lakes. Nevertheless, it is noted that some of the highest fish catches in some lakes (OXB06, BOS10, and OXB05) were recorded during the dry periods when the southern oscillation index was negative. However, long-term trend assessment indicated strong declines in these lakes, which may be a result of other factors. Storey *et al* (2009 a, b) suggested that there is now a change in community structure in some of the lakes downstream of the mining outfall.

#### **3.4.2.4 Mine influence**

While in earlier years all changes in fish assemblage structure of floodplain sites were attributed to natural events and climatic phenomena (Swales *et al*, 1999), the possible contribution of the mine to fish declines is now more certain in mine-impacted lakes (e.g. OXB06, BOS10, OXB05) (Hortle & Storey, 2006). The impacts of the Ok Tedi mine on species diversity and abundance in some of these lakes have also been suspected according to a recent report by Storey *et al* (2009). That report noted that the highest rates of decline in fish species diversity within the floodplain sites were from Lake Pangua (OXB05), Bosset Lagoon (BOS10), Sembe Oxbow (OXB03), and Kwem oxbow (OXB06), all downstream of the mine. This observation is not local but global. Daniel *et al* (2014) showed that the impact of mining on fish communities in large catchments was far greater compared to other anthropogenic activities such as agriculture or urban land use. Reduced fish and macroinvertebrate survival and production have been found in streams receiving mine drainage (Letterman & Mitsch, 1978; Howells *et al*, 1983). The accumulation of both influences from anthropogenic land use plus the unique effects from mines reduces fish species richness and abundance (Letterman & Mitsch, 1978; Howells *et al*, 1983; Ferreri *et al*, 2004; Schorr & Backer, 2006; US EPA, 2011).

When data associated with the *El Niño* events (outliers) were removed from the dataset the regression of the catches over time showed that *N. papuensis* catches declined in OXB06 and BOS10 while catches at OXB05 also showed a negative (but non-significant) trend. This trend is most likely to be mine-related and not due to fishery activity as *N. papuensis* is not a target species in this region of the Fly River, especially as there is no market.

Various suggestions have been postulated as to the reasons for these declines in association with mining activities. The chief suspect has been the gradual habitat destruction through sedimentation. The consequence of increased sedimentation has been that the main channel river bed has aggraded to a point where it is causing an increase in the frequency and duration of inflow to the oxbow lakes and prolonged overbank flooding, resulting in the loss of riparian vegetation and large areas of forest dieback. Suspended sediment levels have also increased in the water column along with an increase in concentrations of several dissolved metals (WRM, 2005) which are toxic or potentially toxic to aquatic fauna. It is not clear how these impacts interact (*cf.* Elliott *et al.*, 2003).

The aggradation of the river bed most likely affects floodplain habitats through overbank flooding, inputs of mine-derived sediments, and die-back of flood-intolerant vegetation (Pickup & Cui, 2009). Recent analysis has also detected a change in the algal contribution to food web of OXB06, which may be related to dieback and sediment inputs (Storey & Yarrao, 2009). Recent investigations have also revealed that acid rock drainage (ARD) could affect floodplain (and riverine) habitats as mine-derived sediments become exposed on levee banks and tie-channel banks of the middle Fly region (Bolton *et al.*, 2009; Apte, 2009). This happens when these sediments undergo oxidation (exposure to air) and subsequent wetting and leaching lead to the release of metals, such as copper, into adjacent surface waters. Fish are sensitive to areas of high metal concentrations (including copper), and avoid them (Sorrenson, 1991). Many fish species (both resident and migratory) would, as a result, avoid areas that are leaching metals, thereby affecting the fisheries ecology of the water body. Further, Newman & Unger (2003) pointed out that in the presence of pollutants, aquatic fauna spend much of their energy on the production of defence proteins (such as cytochrome P-450) with little energy allocated for growth and reproduction, therefore impacting on the next generation. Various other workers have also noted that such responses can produce significant changes in population demographics (Sibly & Calow, 1989; Holloway *et al.*, 1990; Sibly, 1996)

There are, therefore, implications for these impacts on *N. papuensis* populations in each water body. Habitat type and environmental degradation of the habitat both seem to have a compounding impact on the current trends as seen in the spatial distribution of *N. papuensis* in the Fly River lakes especially in OXB02 and OXB06. It is acknowledged, however, that even in the short term, fish abundance can be very variable as could be seen in the study data collected in 2008 and 2009. The mine impacts are possibly not as pronounced yet in the grassed floodplain

sites, although water quality data (Chapter 2) shows that OXB05 is already showing impacts of mine-derived effluents affecting its habitat quality.

### **3.5 Conclusion**

Most of the work undertaken by OTML in the past involved the analysis of total biomass of all species and trends overtime, and information on *N. papuensis* was limited. This study has shown that this species has a substantial influence on these trends due to its large contribution to the total fish biomass. Thus, the fluctuations observed in the fish biomass of the Fly River system is influenced in a large way by *N. papuensis* biomass, and any negative impacts on the biology of this species may indicate a system-wide impact on its predators such as barramundi and black bass. A decline in its biomass would also mean that nutrient recycling in lakes would be impacted because of its habit of feeding on detritus. In addition, its decline in the system would reduce food security of the Fly River people.

This work has shown that *N. papuensis* is more abundant in the grassed floodplain lakes than in the forested lakes, which maybe an indication of poorer water quality closer to the mine and/or it may be a reflection of habitat-influenced differences in resources. Trends in its population in mine-impacted lakes have been negative in the long term probably due to impacts from the mine and not from short-term effects of droughts as previously thought.

It is likely that with uncontrolled and increased levels of sedimentation and pollution by xenobiotics, in addition to climate change, the fisheries ecology of this system is facing serious decline. While a projected increase in rainfall of 10-20% by 2021 and, 5-20% by 2035 and increased river discharge of 9% in the Fly River region (Lough *et al*, 2011; Gehrke *et al*, 2012) might mitigate some impacts, it also enhances the risk of increased sediment and metal deposition in the floodplain of with consequential impacts on the complex ecology and food webs of this system.

## Chapter 4 Variations in *Nematalosa papuensis* population structure

### 4.1 Introduction

Fisheries associated with large tropical freshwater systems support millions of people (Bayley, 1988; Lowe-McConnell, 1996; McGrath *et al.*, 1998; Coomes *et al.*, 2010); however, over-fishing and environmental pressures pose serious threats to the survival and maintenance of these systems (Agostinho, 1985; Agostinho, 1990; Lowe-McConnell, 1996; Adeyamo, 2003). Knowledge of fish biology, population structure, and dynamics is essential if natural fisheries resources of tropical freshwater systems are to be exploited sustainably (Agostinho *et al.*, 1990; Holzbach *et al.*, 2009).

Current understanding of fish populations in tropical rivers and lakes have come mainly from work in South America and Africa, with some additions from Asia. These studies have added to a pool of information on length/weight relationships (Araya *et al.*, 2008; de Souza *et al.*, 2008; Abowei, 2009; Holzbach *et al.*, 2009; Isa *et al.*, 2010), sex ratios (de Souza *et al.*, 2008; Holzbach *et al.*, 2009), and age/size distribution (Bhatt *et al.*, 2000; Araya *et al.*, 2008; Holzbach *et al.*, 2009). Bishop *et al.* (2001) have made some significant contributions to understanding the population structures of tropical species in the western Pacific through work on 40 species in the Alligator Rivers region of northern Australia, of which 35 species are shared with the Fly River region (Allen *et al.*, 2008). In comparison, very few such studies on aspect of life history have been undertaken in PNG, although there are more than 300 freshwater species recorded there (Allen, 1991). These studies include research on the barramundi (*Lates calcarifer*) in the Fly River (Reynolds, 1978; Reynolds & Moore, 1982), and the others in the Sepik, on six catfish species (Coates, 1988, 1991), the tarpon (*Megalops cyprinoides*) (Coates, 1987), one rainbow fish species (*Glossolepis multisquamatus*) (Coates, 1991), the sleepy cod (*Oxyeleotris heterodon*) and the cardinal fish (*Glossamia gjellerupi*) (van Zwieten, 1995).

In addition, very little information is available in the mainstream literature addressing population structure of primary consumers, such as clupeids in floodplain lakes of large tropical river systems, even though they form an important link between the primary producers and the higher-order consumers. This issue is even more crucial for ecosystems such as the Fly, where significant changes have occurred to habitat and fisheries ecology as a consequence of environmental and anthropogenic pressures from mining, and the demands for protein for food security. In fact, fish catches in Fly River have been declining for the last 20 years in both the river channel and the floodplain lakes (Swales *et al.*, 1998; Storey *et al.*, 2009; Chapter 3). There

is, therefore, a need to understand the current changes in this impacted river system at the different levels of biological organisation as well as at different trophic levels.

As seen in Chapter 3, *N. papuensis* is a species with the potential to indicate the ecological health of a river system because of its abundance. Its abundance and population structure in different habitats should be able to reveal not only information about its biology, but that of the health of its habitat and the ecosystem as a whole. This is because the structure and dynamics of a fish community depend on both abiotic and biotic parameters that are responsible for maintaining the ecology of an ecosystem (Vannote *et al.*, 1980; Schlosser, 1982; Junk *et al.*, 1989; Thorp & Delong, 1994; Welcomme & Cowx, 1998). Important abiotic parameters that govern fish ecology include water temperature, oxygen content, available habitats, and size of the water body (Chapter 2). Species composition and densities, food availability, and primary production are among the important biotic factors (e.g. Degani *et al.*, 1998).

Because of the key trophic role of primary consumers, such as *N. papuensis*, it is difficult to establish and manage fisheries based on large species at the top of the food chain – such as barramundi (*Lates calcarifer*) or black bass (*Lutjanus goldiei*) – without understanding the responses of primary consumers to biotic and abiotic factors. In addition, an understanding of the population structure of *N. papuensis* will provide options for the management of any fisheries proposed for this species.

Therefore to achieve the third objective of this thesis, this chapter describes the dynamics of the population structure of *N. papuensis* by investigating its length-weight relationships, sex and size distributions, growth, and juvenile population structure under current Fly River conditions.

## **4.2 Methods**

### **4.2.1 Fish sampling and processing**

All fish samples were collected as described in Chapter 3, with the larger fish (70 - >280 mm) collected by gillnets, and juveniles (< 70 mm) by the use of rotenone. Otolith samples were collected from pond-raised fish in order to cross check with wild fish of same size for age confirmation. Processing for length and weight were as described in the previous chapter.



## 4.2.2 Age and growth determination

### *Examination of otoliths*

Ten specimens of larger fish from a range of size classes (70 – 280 mm; Appendix 4.1) from each lake were preserved and transported to the laboratory on each trip for extraction of otoliths for age determination. The growth of scales and otoliths of fish depends upon environmental conditions and fish physiology. Generally, periods of slower growth of the fish lead to checks (interruptions) in the growth of otoliths and scales. In temperate fishes, major checks occur at annual intervals, usually associated with reduced growth during winter, so checks can be used to estimate the age of fish (Holden & Raitt, 1974). In tropical fishes, checks may occur at variable frequencies and for a range of reasons, which makes the use of otoliths or scales for age estimation of many tropical species quite challenging or difficult (Holden & Raitt, 1974). Daily increments are often visible on younger fish, because the rate at which bony tissues are formed varies diurnally (e.g. Geffen, 1982)

### *Age estimation*

The microstructure of whole otoliths collected from fish from the Fly River lakes was not visible under any magnification, so 37 otolith samples from fish of 83 - 280 mm LCF were prepared by grinding to achieve transverse sections. A compound microscope at 40x magnification was used to count checks. The examination of otoliths showed between 3 and 14 checks per otolith, which could not be age checks, but may be related to changes in somatic growth of the fish or a combination of both, as noted by Hortle & Storey (2006). This was affirmed by the fact that some fish of 70 mm LCF sampled from the Fly River and stocked into fish ponds showed only 1 check on the otolith indicating that the fish is either only one year old or has undergone a first spawning run. This therefore rendered this method unsuitable for the estimation of *N.papuensis* age from otolith checks without prior knowledge of their birth date.

### *Cohort examination and growth estimation*

The examination of cohorts from wild fish was undertaken to see if clear cohorts existed for growth rate estimation. This was done by tracking clear peaks in cohorts from every trip since the first sampling— a method employed by Puckridge & Walker (1990) in the Murray –Darling, and Bishop *et al* (2001) in Northern Australia for *N.erebi*. Changes in fish length were also monitored in pond fish between June 2007 and March 2009.

### *Otoliths from pond fish*

One hundred young *N. papuensis* of approximately 70 mm LCF were seine-netted from the Fly River at Kiunga and stocked in the Provincial Fisheries ponds at Samagos in June 2007.

Phytoplankton from other ponds were introduced into the herring ponds a month prior to the stocking of the herrings. Water flow into the ponds was continuous and came from a tributary stream of the Fly. No fish feed was used to feed the herrings as this could result in altered growth rates and therefore not be representative of the natural conditions. The fish underwent a number of spawning events during this time, and there appeared to be an overlapping of spawning events as they bred continuously throughout the year. The fingerlings from each new spawning event were transferred into new ponds in order to keep track of the cohorts. Leaks to ponds caused the loss of most of the fish; however, 25 individuals from different cohorts survived and were collected during 4 sampling trips to provide otolith and growth data.

Age estimation and growth rates of *N. papuensis* were made from the 25 specimens sampled from the ponds, assuming that fish in the ponds breed and grow at the same rate as in the wild, and that the specimens sampled are representative of the growth and aging of *N. papuensis* in the Fly. Examination of otoliths collected from these fish was undertaken using the methods described previously. The number of otolith checks were then related to the known age of the fish in the ponds.

## **4.2.3 Data analysis**

### **4.2.3.1 Length-weight relationship**

The length-weight relationships (LWR) of all fish, of the form  $W = aL^b$ , (where W = weight; a = y-intercept; L = length; and b = growth exponent or slope) (Pauly, 1983) were log transformed to give straight-line equations of the form:  $\text{Log Weight} = \text{Log } a + b \times \text{Log Length}$ , for each population sample and sex from the different water bodies. The slopes of the lines were tested for similarity using ANCOVA with fish length as the covariate. All analyses were done using Statistica 9.

### **4.2.3.2 Size structure**

Comparison of sex distributions in all lakes at different seasons was undertaken using Log-linear analysis. Log-linear-analysis is another way of examination of cross tabulation tables (to explore the data or verify specific hypotheses), and it is sometimes considered an equivalent of ANOVA for frequency data. Specifically, it makes it possible to test the different factors that are used in

the cross tabulation (e.g., sex, lakes, seasons) and their interactions for statistical significance. Frequency counts of each sex were used as the dependent variable, and lake, season, and sex were used as factors.

A multivariate regression tree analysis was employed to determine the structure of populations in the water bodies during different seasons using frequencies observed in each size class. This method can accommodate a broad range of data and models; it can display a wide variety of interactions; it is able to capture non-additive behaviour; it is unaffected by unbalanced data; it caters for missing values; and it is able to analyse complex multilevel interactions. All analyses were undertaken using Statistica 9 package.

Samples collected from rotenone sampling and other individuals  $\leq 80$  mm were analysed separately as juveniles, based on the assumption that they reach their adults stages around this size class (Medeiros & Arthington, 2008). Those in size classes from 95-275 mm were treated as adults.

#### **4.2.3.3 Age and Growth**

Using the formula  $Growth\ rate = (L_{t_2} - L_{t_1}) / (t_2 - t_1)$ , where  $L_t$  = length, and  $t$  = time (months), fish growth rates were estimated from fish length data collected from ponds using the initial approximate size of 70 mm. The change in length was calculated as the difference between the mean lengths of fish sampled during each new sampling and the previous trip. Change in time was similarly calculated as the difference in months between the previous and new sampling trip. Preliminary analysis showed that growth rate of pond fish and fish in the Fly River fell within the same range, so it was assumed that the growth in the ponds depicted the actual growth in the natural environment, and these rates were then used to compare with growth rates estimated from cohort tracking in three representative lakes with clear cohorts (OXB06, BOS10, and DAV01). Cohorts of fish born in other ponds, that were used to monitor later generations, could not be accurately followed. This is because of the loss of fish due to leaking and drying up of a number of ponds during the study period. Consequently, the age of the fish were then estimated from the equation derived from the plot of estimated age (independent variable) and length (dependent variable). This was estimated from the growth of fish in the pond which had the original stock (ones captured from Fly River) being followed through to the end of the study period, based on the assumption that 70 mm fish were age class 0+. This information was then used to estimate the age of size classes in the natural populations.

## 4.3 Results

### 4.3.1 Length-Weight relationship

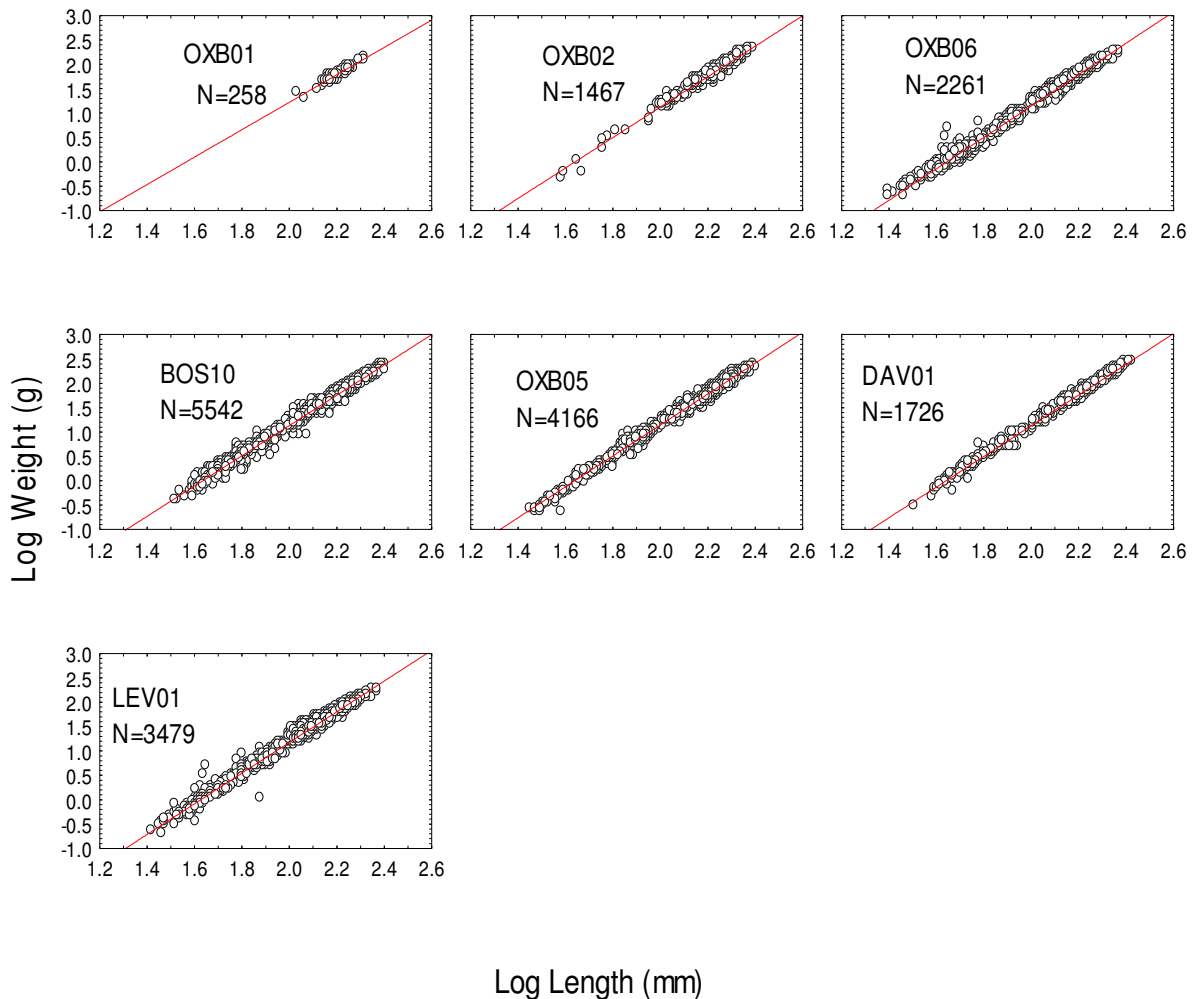
#### 4.3.1.1 Slopes and growth condition

The log-transformed data show very strong linear relationships between fish length and weight (Table 4.1, Fig. 4.1). Due to the low number of surviving fish in the ponds, only a combined LWR could be established, but this shows that pond fish had growth parameters that were within the range found in fish sampled from the wild populations (Table 4.1).

**Table 4.1 Properties of length-weight regression of male and female *N. papuensis* at all lakes. The LWR of pond fish is also included.**

Lake	Sex	Parameter			
		N	Intercept	Slope	r
OXB01	Combined	258	-4.404	2.814	0.942
	M	175	-5.013	3.087	0.985
	F	80	-4.499	2.854	0.982
OXB02	Combined	1,467	-5.126	3.125	0.985
	M	61	-4.886	3.030	0.990
	F	61	-4.689	2.936	0.986
OXB06	Combined	2,261	-5.300	3.223	0.996
	M	63	-5.094	3.121	0.991
	F	38	-4.797	2.983	0.993
BOS10	Combined	5,542	-5.078	3.107	0.996
	M	117	-5.178	3.168	0.991
	F	123	-4.804	3.003	0.969
OXB05	Combined	4,166	-5.198	3.172	0.995
	M	41	-5.067	3.122	0.991
	F	115	-5.156	3.157	0.988
DAV01	Combined	1,726	-5.166	3.149	0.995
	M	122	-5.294	3.229	0.994
	F	215	-5.174	3.176	0.989
LEV01	Combined	3,479	-5.119	3.148	0.993
	M	44	-5.109	3.122	0.985
	F	70	-5.232	3.176	0.983
POND FISH	Combined	25	-4.9156	3.039	0.993

LWRs varied between lakes and sexes. Comparison of combined slopes of both males and females using ANCOVA, with Log length as covariate and lakes (independent variable) and sex (dependent variable) as main factors, showed that there were highly significant lake ( $p < 0.0001$ ) and sex effects ( $p < 0.001$ ). Separate analysis for each sex revealed a weak but significant difference in slope of male LWR between lakes ( $p < 0.05$ ), and a highly significant difference in slope of female LWR ( $p < 0.0001$ ) between lakes (Table 4.2).



**Figure 4.1** Log-transformed linear regressions of length-weight of *N. papuensis* in the seven study lakes.

**Table 4.2 Results of ANCOVA of slopes of each sex of *N. papuensis* between lakes using  $\text{Log}_{10}(\text{Length})$  as the covariate.**

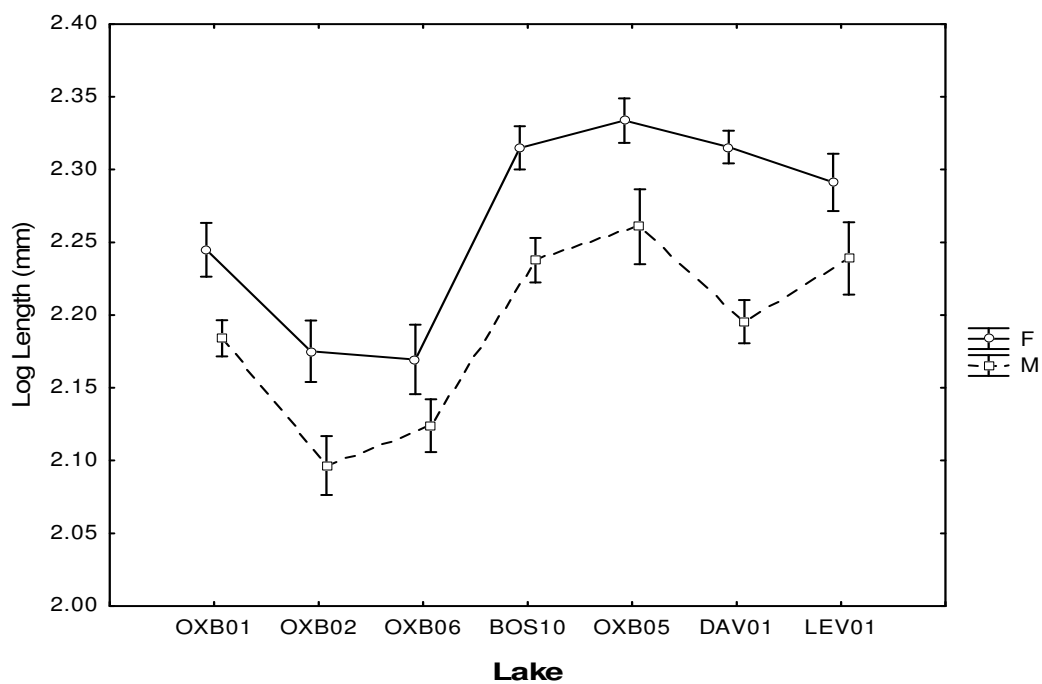
Comparison	Effect	SS	df	MS	F	p
<b>Slopes of Males</b>						
<b>between lakes</b>	Lake	0.018	6	0.003	2.17	0.0447
	Log Lt	32.075	1	32.075	23711.1	<0.0001
	Lake*Log Lt	0.02	6	0.003	2.51	0.0208
	Error	0.859	635	0.001		
<b>Slopes of Females</b>						
<b>between lakes</b>	Lake	0.048	6	0.008	4.76	<0.0001
	Log Lt	29.426	1	29.426	17540.06	<0.0001
	Lake*Log Lt	0.055	6	0.009	5.48	<0.0001
	Error	1.171	698	0.002		

#### **4.3.1.2 Sexual difference**

Results from this study suggest that fish in the forested lakes mature earlier than those found in the mixed vegetation and grass-vegetated lakes. There was a significant interaction, with mean lengths of males consistently smaller than females, at all lakes and substantial differences in lengths among lakes (Table 4.3, and Figure 4.2). Sexually identifiable fish found in the forested lakes fall within the 110-185 mm size range, whilst most in the mixed and grassed floodplain lakes were within the 140 to >200 mm size range.

**Table 4.3 Results of 2-way ANOVA on males and female lengths at all lakes. Log length was used as the independent variable and the factors were Lake and sex.**

Effect	SS	df	MS	F	p
Lake	3.398	6	0.566	80.1	<0.0001
Sex	1.413	1	1.413	200.0	<0.0001
Lake*Sex	0.204	6	0.034	4.8	0.0001
Error	9.521	1347	0.007		



**Figure 4.2 Interaction plot showing the significant interaction ( $F(6, 1347) = 4.7996, P = 0.0001$ ) between sizes of the different sexes and lakes. The lakes are arranged from upstream to downstream. Lines connecting points are to facilitate following of the separate sexes.**

### 4.3.2 Size structure

#### *Population size structures*

##### *Adults*

Male *N. papuensis* predominate in the lower size classes (Table 4.3, Fig. 4.2) whilst the females are more dominant in the larger size classes. Although more females than males were collected during this study (a sex ratio of 1.17 females to 1 male), this difference was not significant ( $\chi^2 > 0.05$  for 1:1 expected ratio).

Log-linear analysis of the marginal and partial associations showed that seasons do not influence sex distribution (Season (2) x Sex (3)); however, when Lake (1) x Sex (3) was dropped from the model the fit became significant. Therefore the final model should include 12, and 23 interactions (Table 4.4a; Appendix 4.1). The results indicate that habitat is responsible for sex distribution. No other interactions provided any improvements to the model (Table 4.4a).

**Table 4.4 a) Log-linear analysis results of comparison of sex for factors: Lake (1) x Season (2) x Sex (3); and b) Marginal Table (Frequency + Delta) for Lake by Sex.**

Factors	Maximum likelihood $\chi^2$	df	p	Delta likelihood $\chi^2$	Delta df	Log-linear fit $\chi^2$
12 13 23	24.336	6	0.00045			
12 13	24.345	7	0.00099	0.009	1	0.924419
12 23	116.51	12	0	92.174	6	$1.07 \times 10^{-17}$
13 23	142.23	12	0	117.894	6	$4.51 \times 10^{-23}$

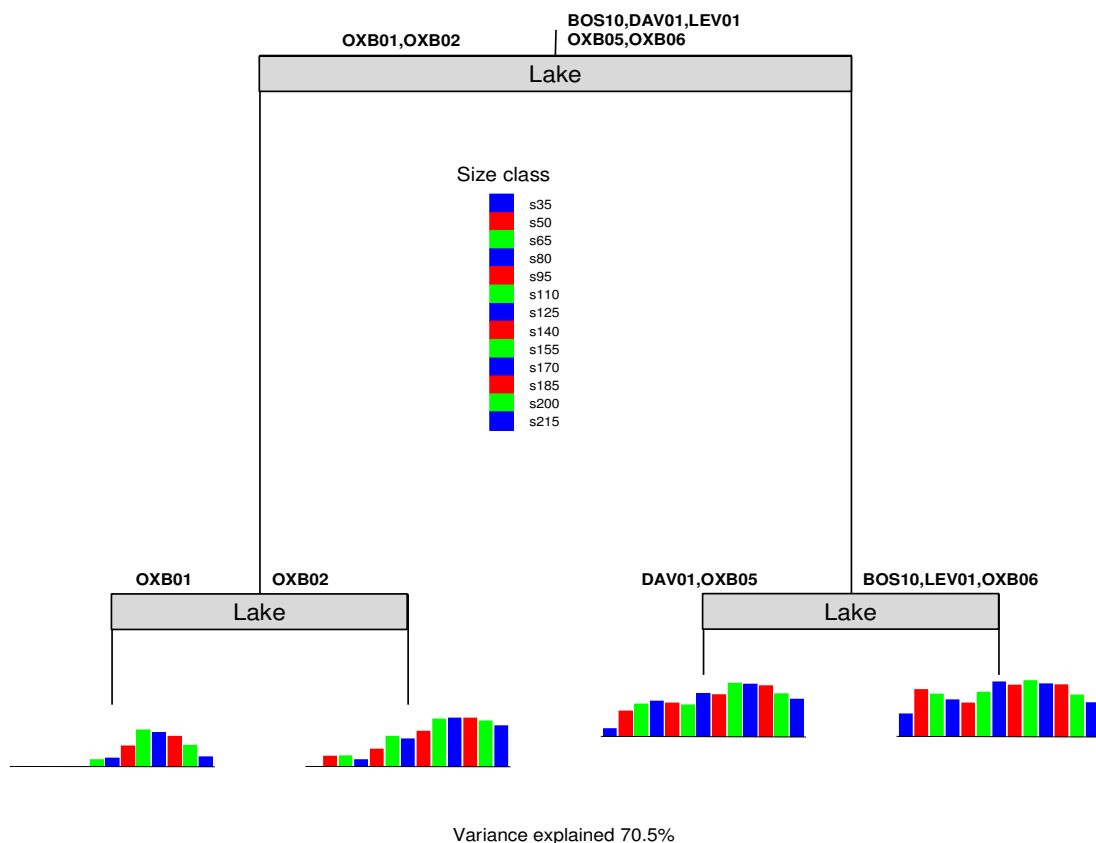
##### **b)**

Sex	OXB01	OXB02	OXB06	BOS10	OXB05	DAV01	LEV01	Total
F	72	58	54	127	99	198	76	684
M	161	67	89	122	42	109	57	647
<b>Total</b>	<b>233</b>	<b>125</b>	<b>143</b>	<b>249</b>	<b>141</b>	<b>307</b>	<b>133</b>	<b>1331</b>



The length size class analysis (of 15 mm intervals) showed that in all the lakes, the majority of the individuals and adults were found in the 110-215 mm size classes, comprising 84.1% of the total population sampled. The juveniles from 35-85 mm made up 13.7 % and those in size classes greater than 215 mm were 2.2% of the population (Appendix 4.1). At all lakes there was evidence of several cohorts present during each sampling trip through the year and during the different seasons (Appendix 4.2).

Multivariate regression tree analysis of the size class distribution showed that there were no clear seasonal differences in size classes; however, there was a significant difference in size class distributions between the lakes (Figure 4.3). The analysis explains 70% of the total variance and shows 4 groups of lakes, with OXB01 and OXB02 having consistently few individuals in the juvenile size classes. A larger percentage of juveniles were found in OXB06, BOS10, and LEV01 on the Strickland.



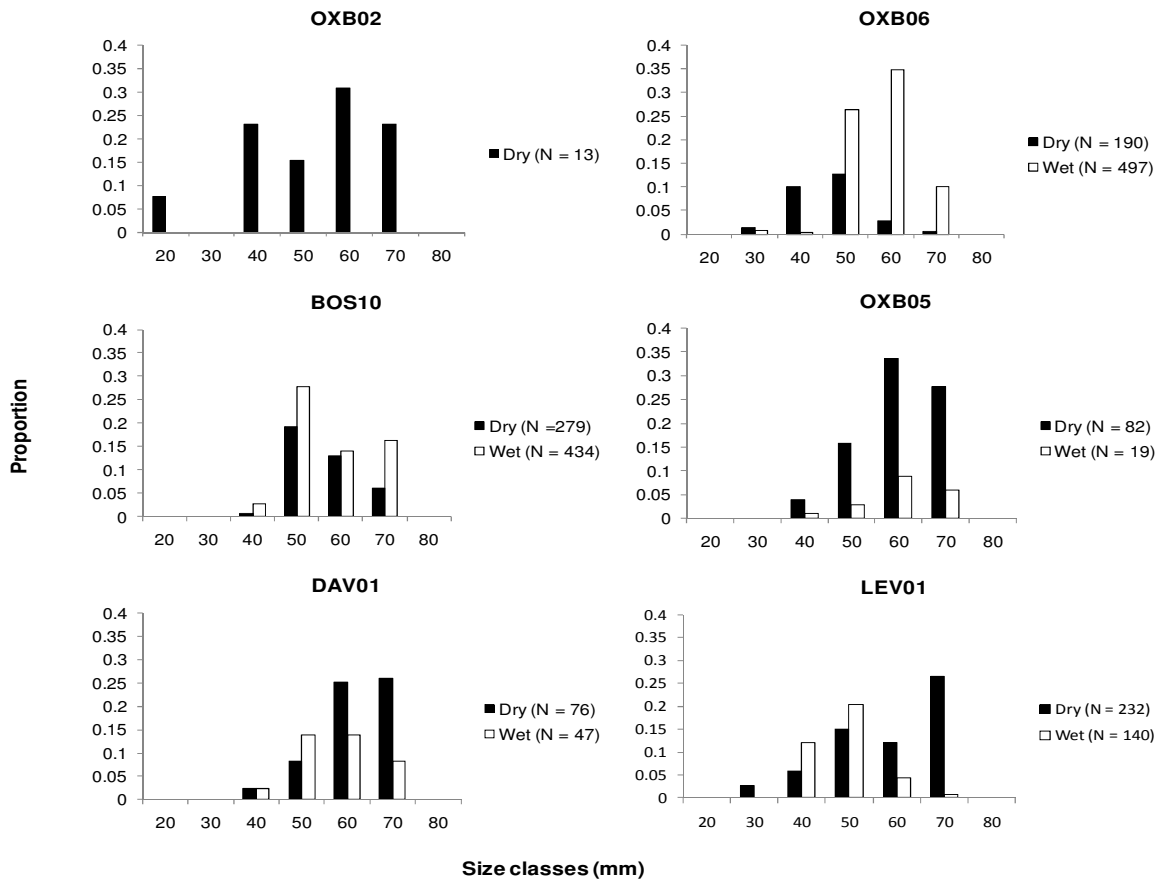
**Figure 4.3** Multivariate Regression Tree result showing the population structure of *N. papuensis* at all lakes.

### *Juveniles*

There were no juveniles collected from OXB01 during this study, and only 13 collected from OXB02. In comparison, the numbers of juveniles collected from OXB06, BOS10, OXB05, DAV01, and LEV01 were 686, 694, 243, 117, and 414 respectively. Size structure showed multiple modes during the year (Figure 4.4). There were more juveniles captured during the wet season at OXB06 and DAV01 while there were more captured during the wet season in the other lakes. Most of them were found within the 50-60 mm size class range (Table 4.5b). The smallest size class collected during this study was 25 mm, from OXB06 and OXB05.

Several cohorts appeared in the different populations (with the exception of OXB01 and OXB02 which did not have sufficient juvenile samples), indicating all year round spawning and recruitment (Figure 4.4). The main modes in the juvenile cohorts appear in the 50-80 mm size classes.

The data showed that the most common juvenile sizes were 50, 60, and 70 mm. Sites OXB06 and BOS10 contributed approximately twice the number of juveniles each to the system as OXB05 and DAV01.



**Figure 4.4 Juvenile *N. papuensis* (below 80 mm in length) size class distribution. No juveniles were caught in OXB01 during the study period.**

### 4.3.3 Age and Growth

#### *Growth*

Tracking of clear cohorts in representative lakes of the Fly (OXB06 for forested lakes, BOS10, and DAV01 for grassed floodplain lakes) during the 2008-2009 study period showed that the mean growth rates for these lakes were 16.03, 13.72, and 10.87 mm/month, for OXB06, BOS10, and DAV01 respectively, putting an average growth rate of fish in the lakes at 13.69 mm/month (Appendix 4.3). A single ANOVA test on this data did not reveal any significant differences in growth rates ( $p > 0.05$ ) between the forested lake (OXB06) and grassed floodplain lakes (BOS10, DAV01), indicating that cohorts in different lakes overlap in growth rates throughout the year with no clear growth rate for each age group.

*N.papuensis* in the ponds grew at an average rate of 11.86 mm/month (Appendix 4.4), which is within the same range as that of estimates based on cohorts in the natural populations. Data in Appendix 4.4 also show a clear decline in average rate of growth with increasing age of fish with growth rates being fastest during the first year of the fish at 25.56 mm/month, lowest at the end of the study at 5.7 mm/month.

### *Age*

Data from pond fish (Table 4.5) clearly show that it is not possible to use otolith or scale checks to estimate the age of *N.papuensis*. The checks on scales and otoliths, in both the wild fish and pond fish, did not appear to indicate annual growth marks. For example, a fish of 162 mm which was a member of the very first cohort spawned in the ponds from Fly River fish was known to be only 2 years old but had 7 checks on its otolith, so it is likely that several checks are formed each year. This may be due to many reasons, such as reaction to changing environmental conditions, dietary shifts, or multiple spawning events.

Based on the monitoring of fish from the ponds between 2007 and 2009, it is assumed that a fish of <90 mm was still less than 1 year old (because of the number of spawning events observed among the original stock), a fish between 90 – 150 mm was 1+, those between 160 – 210 mm were 2+, and those that were >220 mm were 3+ years old. With this information an estimated Von Bertalanffy fitted age-length equation was derived for *N. papuensis* in the Fly River system, where  $L(t) = 434.73 [-\exp (-2.43604*(t-0))]$  (Figure 4.5). Using this equation an estimate was made for the approximate age of *N.papuensis* at different size-classes (Figure 4.6).

Table 4.5 Estimated age range of different size ranges of pond-raised *N. papuensis* and number of checks visible on the otoliths.

Number of fish sampled	Size (mm)	Age (years)	No. of otolith checks
4	70-90	0-1+	1 to 2
5	90-100	1+	2 to 3
4	160-190	1+-2+	6 to 8
4	200-240	2+-3+	9 to 11
4	240-260	3+	10 to 13
4	260-280	4+	13/14

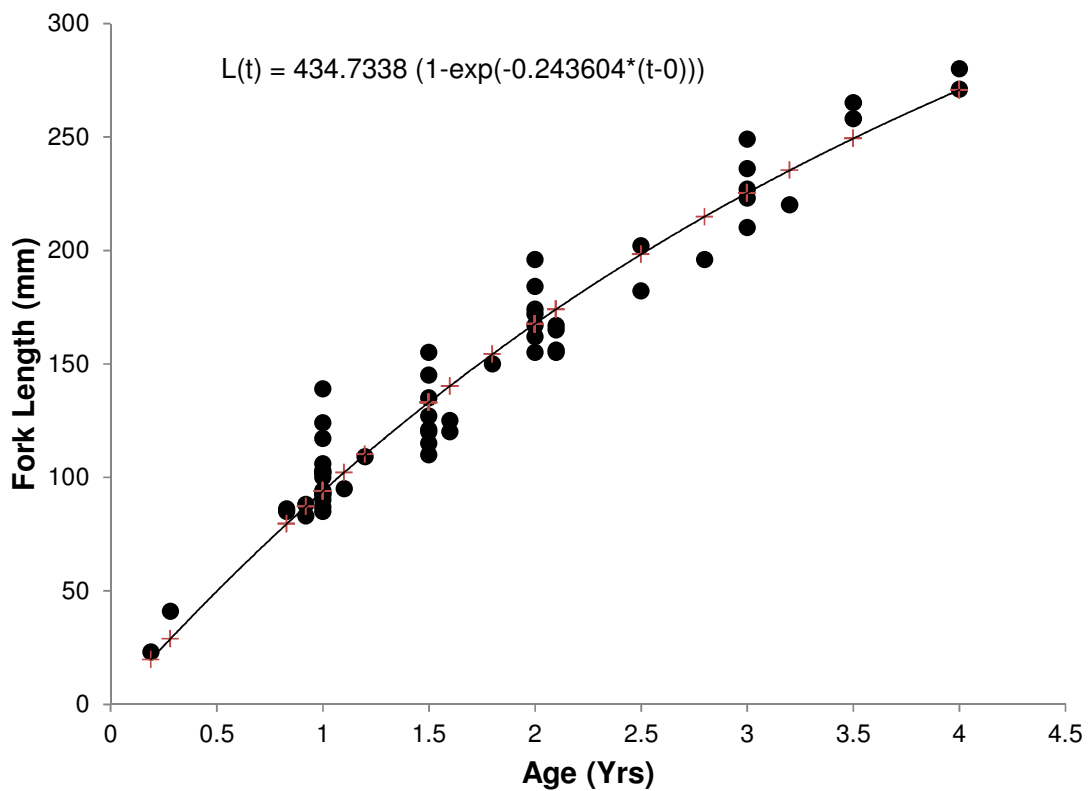
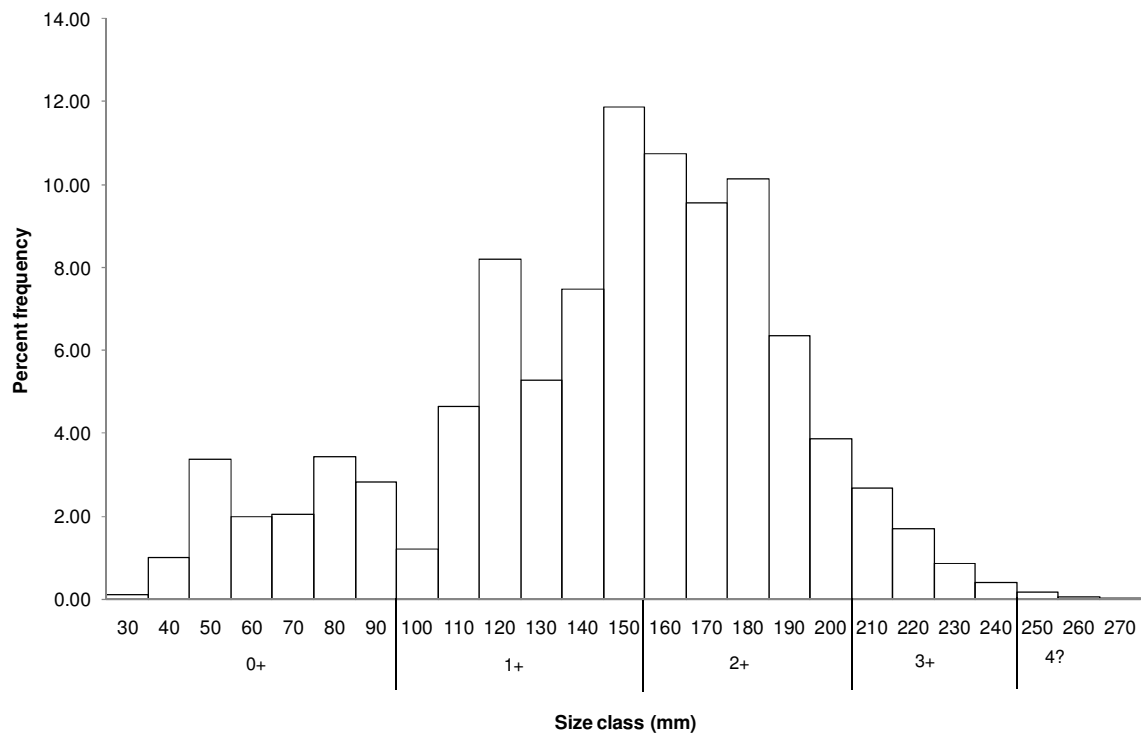


Figure 4.5 A von Bertalanffy fitted age-length growth curve for *N. papuensis* in the Fly River lakes.



**Figure 4.6 Percent frequency distribution of size classes of *N. papuensis* in all Fly River lakes and their estimated age range.**

## 4.4 Discussion

### 4.4.1 Length-weight relationship

Analysis of fish length showed that females of *N. papuensis* are larger than males and that fish from the grassed floodplain lakes were significantly bigger than those found in the forested lakes. This is a trend which was also observed in the Australian *N. erebi*, in which females were more abundant in the larger size classes, and the males dominate the lower size ranges (Puckridge & Walker, 1990; Bishop *et al*, 2001). These results indicate that the LWR of this species cannot be expressed by a single model, but it has to be lake/habitat and sex specific. The role of habitat condition and type in determining fish condition and growth has been reported elsewhere in the tropics such as in Malaysia where 12 freshwater species LWR were studied (Isa *et al*, 2010). Similarly, Kendall *et al* (2009) reported significant differences in growth, mortality, and population structure of Australian mullets in different estuaries. The males and females of these

mullet species clearly demonstrated such differences. Assessment of fishing pressure has utilised length based assessment of population dynamics in sardine fishery in other regions such as Indonesia where populations of similar species were identified as being from separate stocks (Pet *et al*, 1997).

#### **4.4.2 Size structure**

There was no detectable bias in sex ratios. Hortle & Storey (2006) also suggested that the sex ratios are equal, but many small fish which could not be sexed in the field may have been males, and the sampling methods caught mostly larger-sized fish which are mostly females. However, their results were based only on samples from grassland lakes. The current study suggests that more fish in the forested lakes were males, whereas samples from the grassed floodplains were dominated by females. Such differences are known among teleosts and may be a result of various factors such as temperature (Conover & Kynard, 1981), selective mortality by sex through differential predation (Schultz, 1996), and differences in sexual behavior, growth rate, or longevity (Schultz, 1996). Food availability has been identified as another factor that can affect sex ratio in fish populations, with females predominating when there is abundant food, and males predominating where food is limited (Nikolsky, 1963). This suggests that females need better environmental conditions and abundant resources for their development and conditioning to prepare them for spawning events, which in the case of *N. papuensis* is a year-round event.

There was continuous recruitment during the wet and dry seasons, a phenomenon which is not uncommon among lacustrine species when there is abundant food and appropriate conditions all year (e.g., King *et al*, 2003). Similarly, Bishop *et al* (2001) suggested that there is multiple spawning and recruitment in *N. erebi*, in tropical Australia. The results also show that there is a significant difference in frequency of size classes between lakes, indicating that habitat quality and types may determine the numbers and size of juveniles recruited into the populations (*cf.* Hinch *et al*, 1991; King *et al*, 2003; de Melo *et al*, 2009; Neiff *et al*, 2009).

#### **4.4.3 Age and growth**

In temperate Australia, where breeding is seasonal, *N. erebi* growth rates were estimated to be 5 – 6mm/month by tracking size classes through time (Puckridge & Walker, 1990), whereas growth of tropical populations in northern Australian was estimated at 10 – 20 mm/month (Bishop *et al*, 2001), with the higher rate occurring in the summer. Analysis of cohorts by Hortle & Storey (2006) from OTML's historical monitoring program suggests typical growth rates of 25 – 30

mm/month. In this study the mean growth rate inferred from the tracking of cohorts in the wild populations was 13.69 mm/month with a range of 10.87 – 16.03 mm/month. Fish growth rates inferred from stocked ponds fall within the range of 5.7 – 25.56 mm/month. Bishop *et al* (2001) cited data from central Australia that suggested growth of *N.erebi* from 130 – 170 mm over two months – that is a rate of 20 mm/month. The growth rate range for the Fly River *N. papuensis* is similar to that of *N.erebi* in tropical Australia, suggesting a fast-growing tropical species which is a consequence of higher water temperatures and light levels, where the only limit to productivity might be turbidity affecting light levels.

The age-length data from this study using otoliths and tracking of tracking of cohorts will require greater sample numbers, closer tracking of cohorts in a pond situation, or more complex chemical assessment methods to reaffirm the results of this study.

Data from this study and others (OTML pers. comm.; Hortle & Storey, 2006) indicate that *N. papuensis* undergoes several spawning runs each year, causing size-frequency peaks to be obscured as fish in any cohort with faster growth rates overtake the slower growing fish from the previous spawning, so that fish of different ages are found within the same size classes.

In other tropical systems where clupeid fishery was based on multiple spawners there is still an issue with exploitation and management because of unreliable data – for example, in African Lakes (Munyandorero, 2002). In the Indian subcontinent, overfishing was reported in the clupeid *Ilsha ilsha* despite a restriction on net mesh sizes. The issue was mostly to do with enforcement of legislated management plans (Anthony, 1985).

A proper understanding of the biology of *N.papuensis* is needed prior to development of a fishery because in multispawning species different cohorts may have different reproductive fitness. Such has been described by Lowerre-Barbieriet *al* (1998), where it was reported that although early born fish had lower survivorship they produced the most eggs. They found that this was because of their early size at first maturity, low fishing mortality in the first year, and their larger size at age. This finding therefore suggested that multiple spawning could have important implications for recruitment and adult population dynamics, and therefore fishery management.

#### **4.5 Conclusion**

An understanding of the population structure, age and growth of this species is important firstly because it comprises approximately 40 % of the Fly River fish biomass and is a major component



in the diet of predatory species of the Fly River system (OTML, 1995); and, secondly, any sustainable fishery that may be planned for this species can only be successful with a knowledge of its sex, growth rate, size, and population structure in the different habitats. Climatic and sedimentation models show that the Fly River system would be significantly affected by habitat changes therefore this knowledge maybe even more useful, as an indicator of ecosystem health, as populations can be restructured in response to anthropogenic or climatic conditions in relation to impacts on the habitat.

This study has shown that habitat type plays an important role in the abundance and population structure of *N.papuensis*. It has also shown that females of this species are consistently larger than males and are relatively more abundant in grassland than forested lakes. Despite this difference it is encouraging to note that in all habitats juveniles were still being recruited year round into the adult populations.

The result of this study has also shown that *N. papuensis* is a fast growing species, with cohorts in lakes overlapping in growth rates throughout the year. Further studies on the aging of this species may still have to be done to establish a procedure for aging of this species since it was not possible to age it using otoliths.

In view of the finding that habitat type is important in the structuring of *N.papuensis* populations, it is important to note that any negative impacts on habitat quality would significantly affect its populations and, therefore, fish community structure, given that it is the dominant primary consumer. Secondly, for any future consideration of rehabilitation of mine-impacted lakes it is important that viable populations, not only of *N.papuensis* but also of other species, in the un-impacted grassed floodplain lakes be considered for protection and management as breeding and nursery grounds. Further studies would be needed to establish whether the growth rate of this species has been affected by pollution in mine-impacted lakes compared to those in unimpacted lakes.

## Chapter 5 Spatial and temporal patterns in *Nematolosa papuensis* diet

### 5.1 Introduction

In many tropical floodplain river systems seasonal inundation (flood pulse) and a subsequent drying phase are the driving force behind ecosystem function; however, systematic and detailed studies on how hydrology and physico-chemistry affect the flora and fauna in these systems are rare (Junk & Wantzen, 2003). Three ecological models currently attempt to explain the sources of energy that support food webs and therefore riverine communities in these systems. These are: the River Continuum Concept (RCC) (Vannote *et al.*, 1980), the Flood Pulse Concept (FPC) (Junk *et al.*, 1989) and the River Productivity Model (RPM) (Thorp & DeLong, 1994). The RCC postulates that downstream communities derive much of their energy from upstream terrestrial input of carbon detritus, which is transported downstream; in larger rivers autochthonous production also has a major role. The FPC proposes a model that caters for large floodplain rivers where there is interaction between river channel and floodplain food webs driven by production in the floodplain rather than material from upstream. Exchange of materials occurs from floodplain to river as flood waters recede. The RPM emphasizes that in-stream primary production (by phytoplankton, aquatic plants, and benthic algae) and local inputs (from riparian vegetation and other organic matter) are the principal sources of carbon.

It is likely that the functioning food webs in the Fly River channel can be explained by a combination of all three concepts depending on what reach (upland forested, floodplain forested or the flat grassed floodplain reach) of the floodplain is being considered (WRM, 2005). In tropical estuaries of Australia, seasonal freshwater flow resulting in connectivity of pools has been reported as being important in regulating carbon sources (Abrantes & Sheaves, 2010). There is evidence, however, that Fly floodplain lagoon food webs, like other similar systems, function separately with limited connectance to riverine food webs (Bunn *et al.*, 1999; Bowles *et al.*, 2001; Storey & Yarrao, 2009).

The diets of the Fly River fish fauna have been reported previously as part of a pre-mining baseline study when Maunsell & Partners (1982) examined the diets of several species occurring in different habitats along the length of the river. Subsequent studies on the Fly River fish diets have been undertaken (OTML, 1986; Kare, 1991); however, the focus was always on higher-order consumers with very little effort put into understanding the diet of species at the base of the food chain, such as *N. papuensis*. This appears to represent a fundamental gap because this species comprises 38% of the biomass of floodplain fish catches (OTML, 1996; Storey *et al.*,

2009a), and is a dominant prey of high order consumers, making it a major driver of the fisheries ecology of the Fly (Hortle & Storey, 2006).

*N. papuensis* is a member of the clupeid subfamily Dorosomatinae, or gizzard shads, which, as their common name implies, utilize a strong gizzard, or muscular portion of the alimentary canal, to grind mud taken from the bottom of estuaries, lake beds and shallow shelf areas, so as to access benthic microalgae (Longhurst, 1971; Schaus & Vanni, 2000; Gido, 2003; Nelson, 2006).

In Australia, the closely related *N. erebi* feeds on detritus and various algal taxa (Bishop *et al*, 2001; Medeiros, 2004; Pusey *et al*, 2004; Balcombe *et al*, 2005; Sternberg *et al*, 2008). Recent studies using stable isotopes ( $^{13}\text{C}$  and  $^{15}\text{N}$ ) have shown that the Fly herring derive their energy mainly from phytoplankton and periphyton (Bunn *et al*, 1999; Bowles *et al*, 2001; WRM, 2006; Storey & Yarrao, 2009) and as a basal feeder this species is a key constituent in the diet of major predatory species such as barramundi (*Lates calcarifer*) (Kare, 1991; Bunn *et al*, 1999; Swales *et al*, 1999), thus providing a critical link to higher trophic levels. These carbon sources have been reported to be the major drivers of floodplain ecosystems elsewhere (Hamilton & Lewis Jr, 1992; Hamilton *et al.*, 1992; Forsberg *et al.*, 1993; Lewis Jr *et al.*, 2001).

Diet changes in the Australian *N. erebi* have been shown to occur between juvenile and adult at approximately 69-70 mm length (Medeiros & Arthington, 2008). In the Fly River, however, a stable isotope study on *N. papuensis* indicated that there was no change in the sources of carbon for this species during its transition from juvenile to adult, and no shift in trophic level (WRM, 2005).

This study on *N. papuensis* diet was undertaken to understand the variability in dietary habits of *N. papuensis* across different ORWBs and seasons. This was achieved by combining information from gut content analysis with information on what is assimilated over time from stable isotope analysis. In this river system no study has been done to distinguish feeding habits of this and other species during different seasons, or to compare gut content and stable isotope data. This study was done also to understand the feeding habits in a broader range of habitats than the WRM (2006) study, which compared control and mine-impacted sites. Given the likely keystone status of this species, a detailed knowledge of its diet is crucial to understanding its ecology and potential responses to environmental change.

## **5.2 Methods**

### **5.2.1 Sampling design**

The sampling sites in the study were chosen to include a range of habitat types and to include five sites downstream of the Ok Tedi mine (OXB02 – Forested OBL, OXB06 – Forested OBL, BOS10 – Mixed grass and forest fringed BVL, OXB05 – Grassed OBL, and DAV01 – Grassed BVL) and two sites not impacted by mine wastes (OXB01– Forested OBL, and LEV01 – Mixed forested and grass fringed OBL). Fish were collected as described in Chapter 3. Twenty fish specimens were randomly selected from each lake during two wet and two dry seasons for processing for guts and stable isotopes (a total of 280 fish).

No food sources were collected for stable isotope analyses due to certain technical and logistical issues. However, background information was available for comparative purposes (WRM (2006).

### **5.2.2 Fish processing**

Guts (gizzards) were removed from fish, frozen and transported to the laboratory for gut content analysis examination. Size classes were selected with reference to length-frequency histograms for *N. papuensis* previously caught by OTML. Unfortunately all of the smaller size classes for use in this study were destroyed when a power outage caused storage freezers to fail. Only samples greater than 70 mm LCF are used in this study as they had been processed earlier.

A piece of the dorso-lateral muscle of approximately 5 g wet weight was removed from each specimen for stable isotope analysis. This amount was taken as about 5 g wet weight which would give approximately 1 g dry weight of flesh.

### **5.2.3 Gut contents processing**

#### **5.2.3.1 Fullness index**

A total of 280 gut samples were collected of which 202 had food in their gizzards while 78 were empty. Gut fullness index (GFI) was measured as the ratio of gut contents weight to body weight (Hyslop, 1980) as:

$$\text{GFI} = [\text{Weight of gizzard contents (g)} / \text{Weight of Fish (g)}] \times 100.$$

### 5.2.3.2 Contents

Only the contents of the gizzards were examined, firstly under a stereo microscope, then under a high-powered microscope (x200). For the latter examination, a sample of the gut contents was mixed with a little water and then stained with Rose Bengal to highlight organic material. A drop of this mixture was then placed on a slide and covered with a glass coverslip. Using random numbers to select numbers from the scale on the microscope stage, three subsamples were then chosen for examination. The total area under the viewing area occupied by food items was assessed. The food items were then identified and the area occupied by that particular food item estimated as a percentage of the total area occupied by all the food items. The percentages from the three replicates were then averaged to give a final percentage for each food item.

The food categories used encompassed all major food items encountered and corresponded to functional food sources (e.g. aquatic plants, detritus/sediments, unidentified plant matter, zooplankton.) based on results of the pilot study of the gizzard contents (Table 5.1). With the assistance of Dr Carrie Preite of James Cook University, filamentous and non-filamentous algal taxa were identified, where possible, to family or genus (Appendix 5.1). Filamentous algae are recognized by the presence of single-celled chains which form long visible filaments. Free-floating phytoplankton, including diatoms, are recognisable by the presence of a cell wall that consists of two overlapping silica cells grouped together. Other unidentifiable algal matter, assumed to be parts of filamentous algae, and/or benthic periphyton were classified as algal cells and grouped alongside filamentous algae. Zooplankton taxa identified in this study are listed in Appendix 5.2. Unrecognisable contents were classified as inorganic or organic matter as noted above, and other plant matter was recognized by the presence of fibers and plant cells.

**Table 5.1 Food category classification**

Category	Description	Sub-category	Description
1	Aquatic plants	1a	Filamentous algae/algal cells
		1b	Phytoplankton/Diatoms
2	Detritus/sediment	2a	Inorganic particulate matter
		2b	Unidentified organic matter
3	Unidentified plant material	3a	Seeds
		3b	Other plant matter
4	Zooplankton		

### 5.2.3.3 Stable Isotopes

The use of stable isotopes enables the tracing of the flow of elements through the food chain thus making it possible to evaluate the relative importance of different energy sources (Pearson *et al*, 2003) over a period of time. In contrast, traditional diet analysis can only show what food items were ingested by fish at one point in time, and does not give information on whether these food items are actually assimilated (Oliviera *et al*, 2006).

### 5.2.4 Laboratory processing of samples

A total of 280 fish muscle samples were collected from 6 of the 7 lakes in this study. Samples from the 7<sup>th</sup> lake, OXB01, including samples prepared for temporal analysis were all destroyed during a freezer malfunction. Only a few representative samples could be used for this analysis, due to cost restrictions. Tissues were oven-dried at 60°C for 48 hours and ground by hand with a mortar and pestle. Four samples were then randomly selected from each lake (total of 24 samples) and weighed to between 0.8-1.7 mg, pelletised in aluminium foil, and sent to the Griffith University Stable Isotope Analysis laboratory for analysis. Samples were oxidised at high temperature and the resultant CO<sub>2</sub> and N<sub>2</sub> were analysed with a continuous flow isotope ratio mass spectrometer (Europa Tracermass and Roboprep, Crewe, U.K.). Ratios of <sup>13</sup>C/<sup>12</sup>C and <sup>15</sup>N/<sup>14</sup>N were expressed as the relative per mil (‰) difference between the sample and conventional standards (PDB carbonate and N<sub>2</sub> in air) where:

$$\delta X = (R_{\text{sample}} / R_{\text{standard}} - 1) \times 1000 \text{ (‰)}$$

where X = <sup>13</sup>C or <sup>15</sup>N and R = <sup>13</sup>C/<sup>12</sup>C or <sup>15</sup>N/<sup>14</sup>N.

Standard QA/QC protocols were applied at all stages of sample analysis within the Stable Isotope Facility at Griffith University.

### 5.2.5 Data analysis

A 3-way full factorial ANOVA was used to test for significant differences in percent composition of the different food categories found in *N. papuensis* during different seasons at each lake. The number of guts with food available for examination and analysis are as listed in Appendix 5.2. For this analysis percent composition of guts by each food category was used as the dependent variable, and the categorical predictors were lake, season, and food category.

A 2-way full factorial ANOVA was used to test for significant relationships in GFI between and within lakes and seasons. A one-way ANOVA employed to investigate spatial patterns in stable isotopes in fish from these lakes.

## **5.3 Results**

### **5.3.1 Gut fullness**

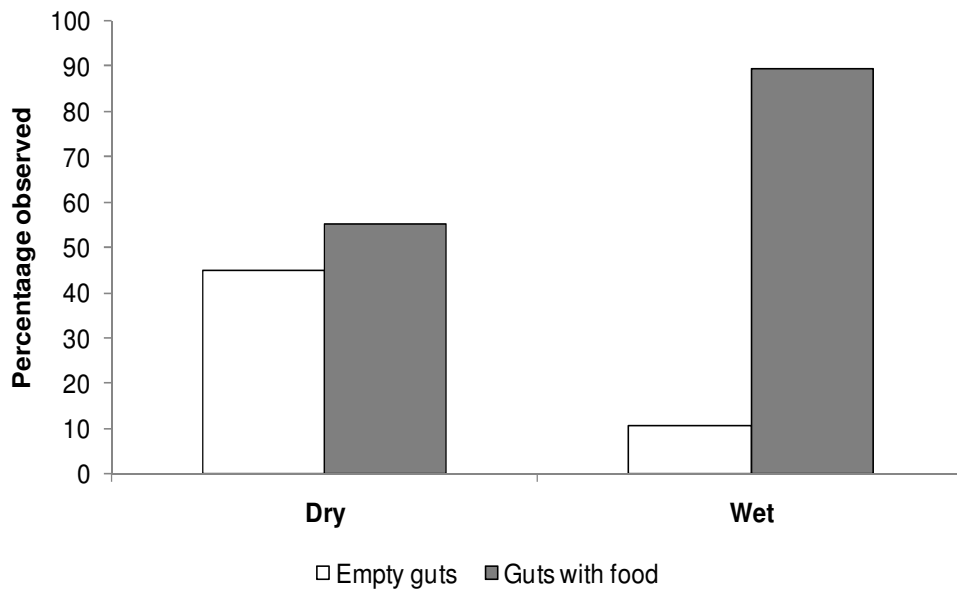
During the dry season the numbers of fish with empty guts and with food in their guts were about equal (ratio of 1:1.2). During the wet season, however, this ratio was 1:8.4. The chi squared test showed a highly significant difference ( $p < 0.0001$ ) between these ratios (Figure 5.1). Mean fullness indices for different seasons and habitat types are summarized in Table 5.2.

The ANOVA showed that there were significant spatial and seasonal differences in gut fullness in *N.papuensis* in the ORWBs (Table 5.3; Figure 5.2). There was also a significant interaction in GFI between lakes and seasons. Overall, the mean GFI was generally higher in the wet season compared to the dry season, and it was higher in the guts from the mixed vegetation and grassland lakes (BOS10, OXB05, DAV01, LEV01) compared to the forested lakes (OXB01, OXB02, OXB06). Fish guts from the mixed vegetation lakes, BOS10 and LEV01 in particular, had higher GFI during the wet season compared to guts from the dry season (Figure 5.2).

**Table 5.2 Gizzard contents (%) and gut fullness index (GFI) in *N. papuensis* from different lakes and seasons. Numbers in parentheses are the number of fish guts out of 20 for each season.**

Food Category		Forested oxbow						Mixed BVL		Grassed oxbow		Grassed BVL		Mixed oxbow		Overall gut occupation by food category	
		OXB01		OXB02		OXB06		BOS10		OXB05		DAV01		LEV01			
Description	Subcategory	Wet (15)	Dry (9)	Wet (17)	Dry (10)	Wet (18)	Dry (6)	Wet (20)	Dry (12)	Wet (18)	Dry (14)	Wet (18)	Dry (15)	Wet (19)	Dry (11)	Total Wet	Total Dry
	1a Filamentous algae/algal cells	14.21	15.78	16.37	16.58	16.88	18.57	19.91	21.58	19.77	18.77	20.65	23.39	21.14	15.03	18.42	18.53
Aquatic plants	1b Phytoplankton/Diatoms	9.11	9.32	7.71	7.83	16.73	8.21	19.64	21.33	21.02	21.13	18.20	19.78	17.12	20.65	15.65	15.46
	2a Inorganic particulate matter	40.62	41.98	36.94	34.23	35.76	30.84	32.01	27.00	27.32	28.63	26.85	23.73	31.31	27.38	32.97	30.54
Detritus/Sediment	2b Unidentified organic matter	32.03	31.80	34.67	37.41	29.47	37.60	27.09	26.92	25.80	28.04	29.69	27.83	28.16	30.64	29.56	31.46
	3a Seeds	2.06	0.33	0.36	0.68	0.00	0.20	0.12	0.00	0.07	0.06	0.18	0.00	0.11	0.00	0.41	0.18
Unidentified plant material	3b Other plant matter	1.69	0.80	2.71	2.24	0.58	3.43	0.60	1.64	5.56	2.17	3.56	4.36	1.82	4.62	2.36	2.75
Zooplankton	4 Zooplankton	0.27	0.00	1.25	1.04	0.59	1.13	0.63	1.53	0.47	1.20	0.86	0.91	0.34	1.68	0.63	1.07
	Percent empty fish	25	55	15	50	10	70		40	10	30	10	25	5	40	10.71	45
	Percent fish with food	75	55	85	50	90	30	100	60	90	70	90	75	95	60	89.29	55
<b>Gizzard fullness index (mean )</b>	Seasonal mean	1.0161	0.975	1.38735	0.997	1.18031	0.72394	1.335685	0.81413	1.41045	1.08	1.6389	1.372	1.473	1.028		
	Lake mean	0.995		1.094		0.866		1.039		1.122		1.371		1.191			

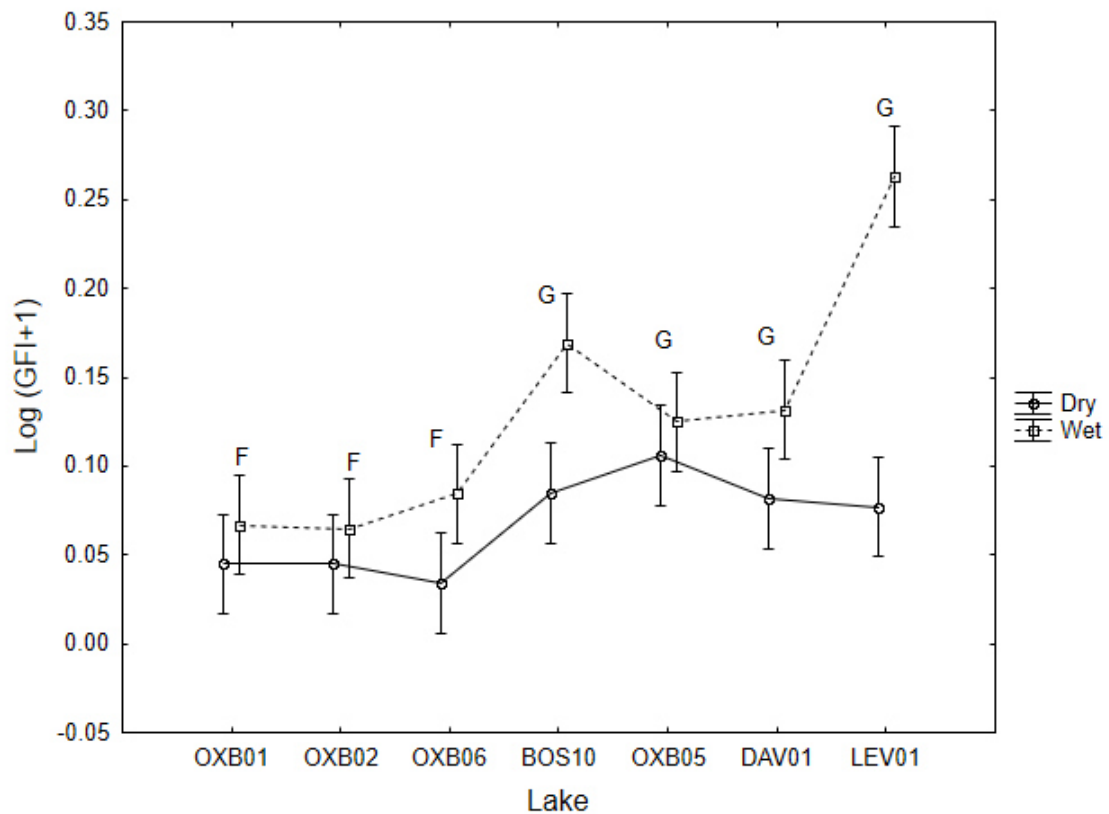




**Figure 5.1 Percentage of fish guts with and without food during the wet and dry seasons. Total number of guts = 280 (140 for each season).**

**Table 5.3 Results of 2-way ANOVA for seasonal variation in GFI at the different lakes.**

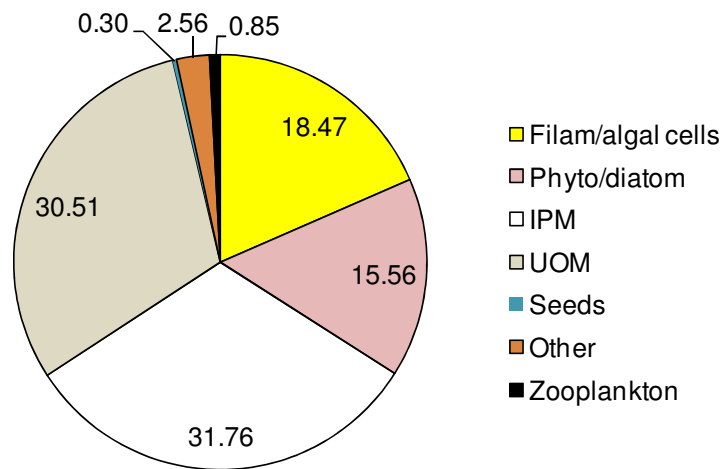
Effect	SS	df	MS	F	p
Lake	0.461	6	0.077	5.655	<0.0001
Season	0.266	1	0.266	16.388	<0.0001
Lake*Season	0.215	6	0.036	1.634	<0.0001
Error	1.075	266	0.004		



**Figure 5.2 Interaction plot of seasonal variations in mean GFI ( $\pm$  95% CI) in the lakes (F (6, 266) = 7.6746,  $p < 0.00001$ ) where F = Forested lake and G = Grassed floodplain lake. Lakes are arranged from upstream to downstream.**

### 5.3.2 Gut contents

The main dietary components were algae, inorganic particulate matter (IPM) and unidentified organic matter (UOM) (Tables 5.2 and 5.5; Figure 5.3). The identifiable microalgae in the gizzard comprised 32 genera with Chlorophyta being represented the most (50%), then Bacillariophyta (25%), Euglenophyta (9.4%), Cyanophyta (6.3%), and 3.1% each for Dinophyta, Cryptophyta and Chrysophyta. The largest percentage of phytoplankton were found in fish from the grassed floodplain lakes (DAV01, 26%; BOS10, 19%; OXB05, 17%; LEV01, 14%) compared to the forested lakes (OXB01, 4%; OXB02, 9%; OXB06, 11%). Detritus (IPM and UOM) content was consistently higher in the gizzards of fish from forested lakes (OXB01, OXB02, and OXB06) than in those from open grassed floodplain lakes. Fish from the open grassed floodplain lakes, on the other hand, had more algal content in their diet than those from the forested lakes. Zooplankton, other plant matter, and plant seeds were minor constituent of the guts.



**Figure 5.3 Percent composition by volume of gizzards of *N. papuensis* from all lakes. Filam/algal cells = Filamentous algae and algal cells; Phyto/diatom = Phytoplankton and diatoms; IPM = Inorganic particulate matter; UOM = Unidentified organic matter; seeds = plant seeds; Other = other plant matter (OPM); and Zooplankton = microcrustaceans and other zooplankton.**

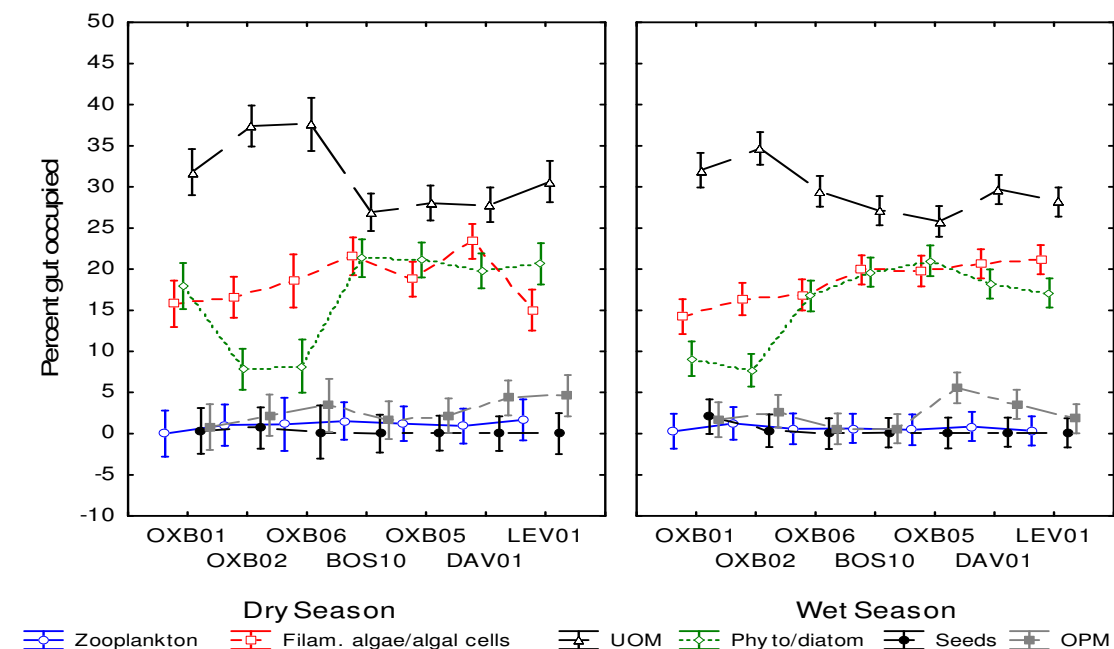
There was a significant three-way interaction between lake, season, and food category for percent occupation of guts by different food categories (Table 5.3). The interaction was driven by complex changes across the large data set (Figure 5.4). Despite the interaction, detritus (IPM and UOM) dominated items found in the guts implying an active consumption of detritus to extract food items. This is followed by the phytoplankton/diatoms and filamentous algae/algal cells groups. The significant interaction between lake, season and food category (Table 5.4; Figure 5.4) showed a clear separation of UOM from all other food categories, being higher in the forested lakes than in the grassed floodplain lakes. The opposite pattern is seen in the amount of gut space occupied by phytoplankton and diatoms, implying that where there is a low availability of phytoplankton, there is a high consumption of detritus, possibly to extract organic matter.

The presence of filamentous algae and fragmented algal cells was generally higher than phytoplankton and diatoms in the fish from forested lakes, and this food category generally increased in the diet downstream from forested to the grassland lakes. Fish in the grassland lakes generally had similar contributions of phytoplankton/diatoms and filamentous algae/algal cells in their diet. These patterns were similar in the wet and dry seasons, indicating a clear habitat influence on their diet. Whilst there was an increasing trend in phytoplankton/diatom content of the diet downstream into the grassland lakes in the wet season, this pattern varied during the dry season in the grassland lakes (Figure 5.4).

Minor dietary items were seeds, OPM, and zooplankton. Seeds may have been consumed opportunistically with other food items, while OPM may be grazed plant matter from periphyton or other macrophytes. The zooplankton content was low, but as zooplankton is easily and quickly digested, even if large amounts were consumed, only a small amount may be seen in the guts.

**Table 5.4 Three-way factorial ANOVA of results of fish diets (food categories) from different lakes and seasons. Dependent variable is percent gut occupied.**

Effect	SS	df	MS	F	p
Lake	233.01	6	38.84	4.773	0.0000
Season	49.20	1	49.2	6.046	0.0001
Food Category	66106.8	5	13221	1624.9	0.00000
Lake*Season	23.75	6	3.96	0.486	0.819
Lake*Food Category	3187.5	30	106.3	13.058	0.00000
Season * Food Category	63.98	5	12.8	1.573	0.166
Lake * Season * Food Category	832.6	30	27.75	3.411	0.00000
Error	4198.6	516	8.14		



**Figure 5.4 Interaction plot showing significant relationships ( $F(30, 516) = 3.4107$ ,  $p < 0.00001$ ) in mean percent gut contents ( $\pm 95\%$  CI) during the wet and dry seasons in each lake. Lakes are arranged in order of increasing distance downstream and from the mine.**

### 5.3.3 Stable isotopes

Stable isotope analysis indicates that there were significant differences in  $\delta^{13}\text{C}$  and  $\delta^{15}\text{N}$  in fish from the different water bodies (Table 5.6; Appendix 5.3). Fish from grassed floodplain lakes (DAV01, OXB05) were clearly more  $\delta^{13}\text{C}$  enriched than those from the forested lakes (OXB02, OXB06) ( $p < 0.0001$ ; Table 5.6; Figures 5.5 & 5.6). There is a clear transition between the carbon signatures of fish from forested (very depleted  $\delta^{13}\text{C}$  values) to the strictly grassed floodplain (relatively enriched  $\delta^{13}\text{C}$  values) lakes. Fish from lakes that have a mixed riparian vegetation of forests and open grass - aquatic macrophytes (BOS10, LEV01) had  $\delta^{13}\text{C}$  values in the intermediate range (Figures 5.5).

The Tukey's tests for spatial variation (Table 5.6) showed that fish from DAV01 had the highest mean  $\delta^{13}\text{C}$  while fish at OXB06 had the lowest. The  $\delta^{13}\text{C}$  in DAV01 fish did not differ significantly from fish in OXB05, LEV01, and BOS10, but was significantly different from that of fish from OXB02 and OXB06. Fish in OXB05 also had  $\delta^{13}\text{C}$  values that differ from fish found in OXB06, but not different to fish from other lakes. Fish from LEV01, BOS10, OXB02, and OXB06 did not differ significantly in their  $\delta^{13}\text{C}$  content. Generally, there was lower mean  $\delta^{13}\text{C}$  content in fish from the forested lakes than in those from grassland lakes.

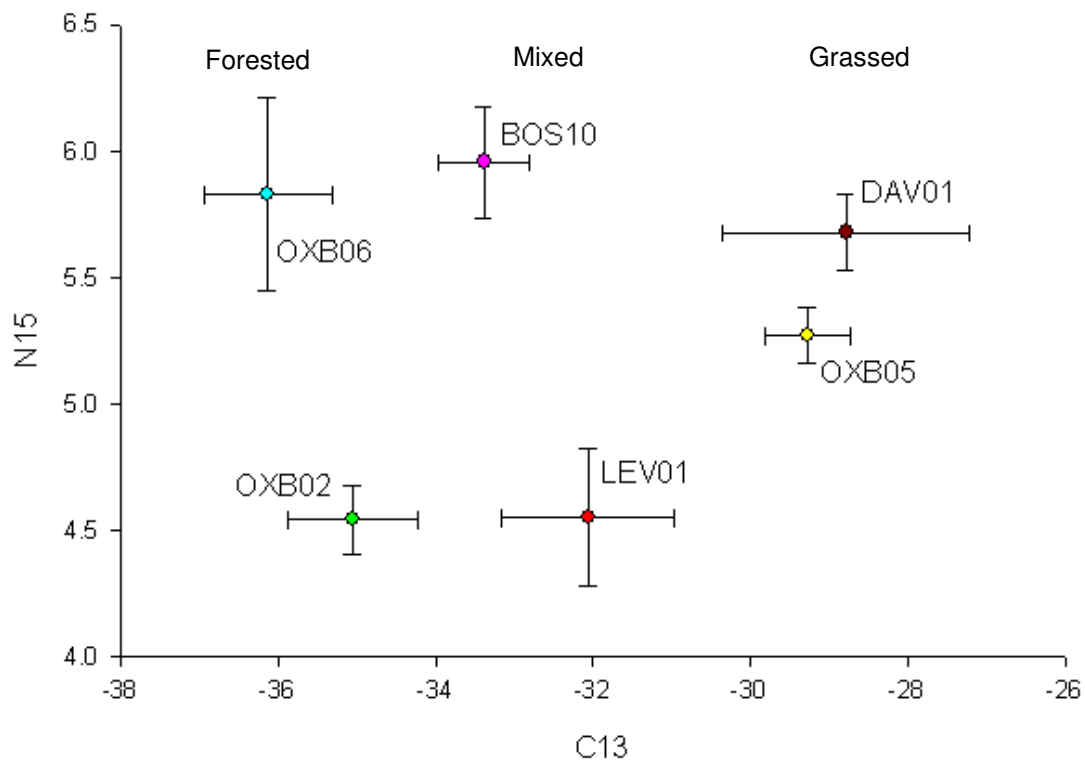
Values of  $\delta^{15}\text{N}$  show that fish from OXB02 and LEV01 were separated from the rest of the group by being the most nitrogen depleted (Table 5.6; Figure 5.5). However, the  $\delta^{15}\text{N}$  signature in these two lakes was not significantly different from OXB05 (Table 5.6). Fish from BOS10, OXB06, DAV01, OXB05, and LEV01 did not show significantly different  $\delta^{15}\text{N}$  signatures from each other. In general,  $\delta^{15}\text{N}$  of *N. papuensis* from grassed floodplain lakes had higher  $\delta^{15}\text{N}$  than those from forested lakes or lakes at higher elevation.

The Tukey's tests for spatial variation in  $\delta^{15}\text{N}$  (Table 5.6) showed that fish from BOS10, DAV01, OXB05, and LEV01 did not differ from each other, but  $\delta^{15}\text{N}$  values in fish from BOS10, OXB06, and DAV01 differed significantly from those of fish from OXB02. Fish from LEV01 and OXB02 had the lowest mean  $\delta^{15}\text{N}$ . Generally there was higher  $\delta^{15}\text{N}$  found in fish from the grassland lakes.

The stable isotope data ( $^{13}\text{C}$  and  $^{15}\text{N}$ ) from this study were plotted against  $^{13}\text{C}$  and  $^{15}\text{N}$  in fish and some producers from some of the same lakes as presented in Figure 5.6 to compare against data from a previous study by WRM in 2005 (WRM, 2006)..

**Table 5.5 One-way factorial ANOVA results and Tukey's significance test for spatial difference in  $\delta^{13}\text{C}$  and  $\delta^{15}\text{N}$  in tissues of *N. papuensis* from all lakes. Lines join lakes that are not significantly different. Results are arranged in order of descending means.**

Effect	SS	df	MS	F	p	Tukey's HSD Test
$\delta^{13}\text{C}$						
Lake	179.76	5	35.95	9.480	0.0001	DAV01 OXB05 LEV01 BOS10 OXB02 OXB06
Error	68.27	18	3.79			
$\delta^{15}\text{N}$						
Lake	7.971	5	1.594	7.482	<0.0006	BOS10 OXB06 DAV01 OXB05 LEV01 OXB02
Error	3.835	18	0.213			



**Figure 5.5 Carbon 13 and Nitrogen 15 signatures in fish at different lakes. The error bars are  $\pm$  1SE of  $^{15}\text{N}$  and  $^{13}\text{C}$ .**

No *N. papuensis* samples were available from OXB01 for this study, but fish collected from this lake in 2006 showed highly depleted  $^{13}\text{C}$  values, with the signature about the same as that found in zooplankton (WRM, 2006). Nitrogen in that lake was also highly depleted as shown by the N

signature in the periphyton, filamentous algae, *Ipomea* and in particulate organic matter (POM) (WRM, 2006). The N signature in *N. papuensis* in this lake was about the same range as that of all the other fish species except *S.kreffti*, which is a top carnivore. Carbon in *N. papuensis* from this study compared to that of 2005 showed more enrichment in OXB02, but more depleted in OXB06, BOS10, and DAV01. The C signatures in the latter 3 lakes are more similar to that of zooplankton, whilst the only phytoplankton sample in the 2005 study at DAV01 had a C signature closer to that of *N. papuensis* found in this study.

Generally, *N. papuensis* had C and N signatures that were closer to phytoplankton, periphyton, zooplankton and particulate organic matter (POM) (Figure 5.6), thereby supporting the view that this species consumes and assimilates these items.

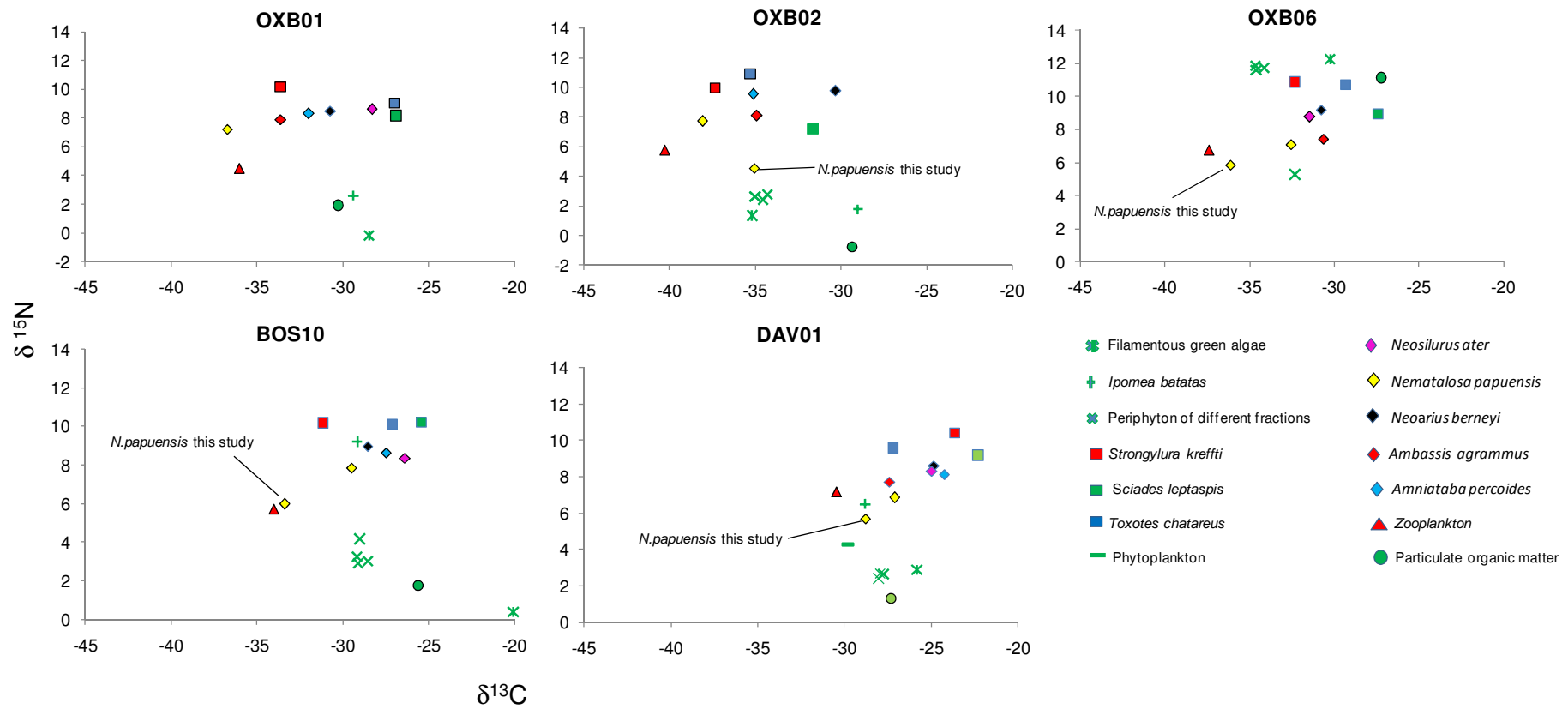


Figure 5.6 Stable isotope signatures of *N. papuensis* sampled during this study compared to those of some fish and energy sources sampled in some of the study lakes in 2005 (data from WRM, 2006).



## 5.4 Discussion

### 5.4.1 Gut fullness

In the Fly River, mean gut fullness index of *N. papuensis* was found to be higher in fish from the mixed vegetation and grass vegetated floodplain lakes compared to fish from the forested lakes. The benthic and littoral zone substrates of the mixed vegetation and grass floodplain lakes of the Fly River are similar in composition to some of the forested lakes (OXB02 and OXB06), but different from others (OXB01), which have more clay texture with leaf litter (Chapter 2). Mean gut fullness in *N. erebi* was found to differ between different habitats in seasonally influenced northern Australian river catchments (Bishop *et al.*, 2001). In the Magela catchment the mean fullness index from habitats with sandy substrates was the highest, while in the Nourlangie catchment, specimens from the escarpment main channel water bodies were found to have the highest indices (Bishop *et al.*, 2001). The observed Fly River patterns were clearly related to greater food availability in open grassed floodplain lakes compared to forested lakes, rather than simply substrate preference. In studies elsewhere it has been noted that floodplain water bodies are more productive and therefore have more food resources for different fish life stages (e.g. Balcombe *et al.*, 2007). These flood plain water bodies become more productive during the wet-season high-flow periods (e.g. King *et al.*, 2003; Balcombe *et al.*, 2007). Mean gut fullness for *N. papuensis* found in this study showed a consistent pattern of higher values during the wet season than the dry season, as found previously for *N. erebi* (Bishop *et al.* 2001).

### 5.4.2 Gut contents

Fine sediment and detritus was an important component of the gut contents of *N. papuensis* in all the lakes and it was strongly correlated with unidentified organic matter, implying that it actively eats sediment, periphyton, seston, and/or possibly grazes surfaces of macrophytes to extract its food, thereby accumulating detrital matter as a consequence. It seems to engage in this habit more in the forested lakes than in the grassed floodplain and mixed vegetation water bodies. On the other hand, there is a shift in feeding habit in the mixed and grassed floodplain lakes where there seem to be equal time/resources spent on consuming detritus as well as planktonic algae, periphyton and filamentous algae. Algae were found to feature prominently in the gut contents of fish from the mixed and grassed floodplain water bodies.

The diet of *N. papuensis* is similar to that of other species in this family, such as the American gizzard shad, *Dorosoma cepedianum*, whose diet was reported to be composed mainly of detritus and algae (Gido, 2003). It is also very similar to that of its Australian relative, *N. erebi*, which in the Australian dry tropics consumed mainly algae, detritus and incidental organic material in main channel water bodies; smaller proportions of algae and detritus and a large component of

microcrustaceans in smaller sandy creeks (especially among juveniles); and about equal proportions of algae (mainly phytoplankton) and detritus with few cladocerans in the backflow billabongs (Bishop *et al.* 2001).

WRM (2007) found that forested lakes of the Fly (OXB01, OXB02, and OXB06) had lower mean phytoplankton taxon richness compared to the grassed floodplain lakes (DAV01 and BOS10). Low phytoplankton biomass in water bodies is indicative of low primary productivity (e.g. Karlsson *et al.*, 2009; Ndebele-Murisa *et al.*, 2010). These forested lakes are probably less productive as a consequence of limited nutrient input (Payne, 1986) and/or light penetration (Karlsson *et al.*, 2009) because of forest cover, resulting in low phytoplankton biomass and other aquatic flora growth as is the case in tropical forested rivers and lakes generally (Payne, 1987; Lewis, 2000). This may also offer an explanation for lower aquatic macrophyte diversity in the Fly River forested lakes. Lack of algae in the open pelagic waters would result in species such as *N. papuensis* resorting to benthic detrital feeding, thereby explaining the high detrital content of guts in the forested lakes. Allen *et al.* (2000) noted that the abundance of gizzard shads in Florida lakes was correlated very closely with the abundance of chlorophyll and lake sizes. The abundance of plankton in the open grassed floodplain lakes of the Fly River would explain why Smith (1998) noted that *N. papuensis* were open-water planktivores. The gut contents of *N. papuensis* therefore reflect its habitat. Similarly, De Longh *et al.* (1983) found that in Lake Kivu, Rwanda, the clupeid *Limnothrissa miodon* was an exclusive planktivore when in open pelagic waters, but resorted to benthic feeding when in the littoral waters.

This pattern of feeding was found to be the same during both the wet and dry seasons implying that there is a very strong location rather than seasonal influence on the availability of food and food types. In the lacustrine environment it is possible that phytodetritivores such as *N. papuensis* do not alter their feeding patterns because of the high abundance of food sources such as plankton. Weliange & Amarasingke (2003), for example, reported that phytodetritivores in Sri Lankan reservoirs showed no significant variation in diet during different seasons. This lack of seasonal variation in diet is also reported among planktivores and phytodetritivores elsewhere in permanent tropical water bodies (e.g., Weliange & Amarasinghe, 2003). This is in clear contrast to *N. erebi* in the Australian seasonal dry tropics, where Bishop *et al.* (2001) reported that late-dry season gut contents of *N. erebi* were based primarily on detritus with a large unidentified component of organic material. There was then an increase in algal component of the diet during the wet season, while detritus remained a large proportion of the diet throughout, but decreased during late wet to early dry season possibly due to anoxic conditions at the bottom, causing them to filter-feed on phytoplankton in the mid-water zones. Stenberg *et al.* (2008) found that *N. erebi* in south-western Queensland were 'boom and bust' feeders that fed more on benthic (non-filamentous) algae during the wet season and filamentous algae and detritus during the dry season.

Conversely, the pattern of less phytoplankton in the diet observed in the forested lakes directly downstream of the mine (OXB02 and OXB06) may also be a result of increased suspended sediments and metals impacting on phytoplankton. These impacts would have resulted in alteration to phytoplankton and other algal assemblages, and possibly loss of some sensitive species rendering them unavailable for planktivores (WRM, 2007; Storey *et al.*, 2009; Stauber *et al.*, 2009). This study also noted the presence of a number of zooplankton taxa (Appendix 5.4), indicating that they may be important in the diet of *N. papuensis*, as they are in *N. erebi* in Australia (Bishop *et al.*, 2001; Medeiros & Arthington, 2008). However, possibly, faster digestion of zooplankton than plant matter would result in a disproportionately low number in the guts.

### 5.4.3 Stable isotopes

#### 5.4.3.1 Carbon

This study shows that the main source of energy for *N. papuensis* is of algal origin, especially from microalgae. WRM (2005) found that carbon in the producers in OXB01, OXB02 and OXB06 ranged from between  $-29\text{‰}$  to  $-35\text{‰}$ , while the carbon found in producers of the grassland lakes ranged from  $-20.8\text{‰}$  to  $-30\text{‰}$ , indicating that the grassed floodplain lakes were more enriched with  $^{13}\text{C}$  (Figure 5.6). Previously Bunn *et al.* (1999) reported that epiphytic and planktonic microalgae were both significantly  $^{13}\text{C}$ -depleted compared to terrestrial sources in the Fly River system. A plot of the mean  $^{13}\text{C}$  in *N. papuensis* from this study against *N. papuensis*, some primary producers, zooplankton, and other fish species sampled from OXB02, OXB06, BOS10 and DAV01 in 2005 (WRM, 2005) reveal that  $^{13}\text{C}$  in fish from forested lakes were more depleted ( $-27$  to  $-40\text{‰}$ ) than in the grassland lakes ( $-20$  to  $-34\text{‰}$ ). This comparison also revealed that *N. papuensis* had a C signature that is close to that of zooplankton in OXB01, OXB06, BOS10, and DAV01. Inconspicuous primary producers are known to play an important role in the food webs of a wide range of aquatic ecosystems, including temperate and tropical wetlands (Hamilton *et al.* 1992, Bunn & Boon, 1993), arid zone rivers (Bunn & Davies 1999), tropical streams (Bunn *et al.* 1999), and coastal mangrove/sea grass systems (Kitting *et al.* 1984; Loneragan *et al.* 1997).

Of the four sources of carbon identified by Bunn *et al.* (1999) in the Fly River system, microalgae were identified as the main contributor to the aquatic food web. By using mixing model (maximum and minimum) estimates for each species and their biomass proportions from OTML historical gill net data, Bunn *et al.* (1999) estimated that algal carbon contributes approximately 32 (minimum) - 72% (maximum) of all energy requirements of fish in the Fly River floodplains. They added that the estimated minimum values may have been very conservative lower estimate, and that in reality, the maximum values were possibly nearer to the actual algal carbon contribution to fish biomass.

Bunn *et al.* (1999) and WRM (2005) showed that there is a very high contribution of microalgae to carbon in Fly River fish, as well as, a significant contribution from terrestrial C3 inputs, but no contribution from C4 plants. These findings were similar to those of Bowles *et al.* (2001) from Lake Murray (PNG) and in other tropical systems such as the Orinoco River in Venezuela (Hamilton *et al.* 1992), Amazon (Forsberg *et al.* 1993), Ohio River (Thorp *et al.* 1998) and Bamboo Creek, Australia (Bunn *et al.* 1997). Both planktonic and epiphytic algae appear to be important at these lakes; however, the high biomass of *N. papuensis* recorded from the Fly sites and their known dominance in the diet of high-order consumers, such as *L. calcarifer*, suggest a greater contribution via this plankton pathway.

#### 5.4.3.2 Nitrogen

The data from WRM (2005) showed that there was an enrichment of N in OXB06, even in the producers at this lake, compared to the other lakes, which is most probably due to forest die-back causing the release of N into that lake. Nitrogen in periphyton, filamentous algae, and POM in all the other lakes was highly depleted.

In this study, fish from the grass floodplain lakes (BOS10, OXB05, DAV01) including OXB06 show higher mean  $\delta^{15}\text{N}$  values compared to OXB02, a forested lake, and LEV01, a mixed vegetation lake, both of which are at higher elevation than the grassed floodplain lakes. This high  $\delta^{15}\text{N}$  in *N. papuensis* of grass floodplain lakes possibly reflect the nature of these water bodies where there is high aquatic macrophyte presence and cover. The decomposition of organic matter can lead to nitrogen transformations which result in elevated  $\delta^{15}\text{N}$  values of primary sources via the equilibria between  $\text{NH}_3$  (gas),  $\text{NH}_3$  (aqueous) and  $\text{NH}_4^+$  (aqueous) (Waser *et al.* 1998), or denitrification within sediments (Fourqurean *et al.* 1997).

The high  $\delta^{15}\text{N}$  in fish from the forested OXB06 seemed to be a disparity; however, it may in fact be reflective of the current status of this water body, as its forested riparian vegetation has died off due to heavy die-back as a result of sedimentation and river bed aggradation (Storey *et al.*, 2009). The death of forests would release nutrients (such as N and P) through run-off or via decaying into the system, which were previously locked up by the plants (Payne, 1987). The low level of N in fish of some of these lakes may be attributed to low nutrient, possibly nitrogen limitation, and therefore the nutrient status and productivity in these lakes (e.g., Douglas *et al.*, 2005).

### 5.5 Conclusion

This work has provided a clearer understanding of the short-term dietary habits and components of *N. papuensis*, establishing the fact that its main carbon source is algal-based. This study has also for the first time established an insight into the foraging habits of this species in the different habitats of the Fly River ORWB systems, showing that it is more a pelagic feeder in the grassed

floodplain lakes and resorts to detritus shifting in the forested lakes where planktonic food sources are limited.

In addition, the results of this study indicate that there is more food available during the wet than dry season, which is reflected by fuller guts in the wet season, and secondly, this species does not change its choice of food seasonally unlike the Australian *N.erebi* (Stenberg *et al*, 2008). The implication of this dietary pattern is that if the water quality of the Fly River lakes become seriously affected by mine wastes through acidification and TSS, thereby affecting primary production through either alteration of, or depletion of phytoplankton populations in the Fly River lakes; as well as long term combination of both mine pollution and climate influenced alterations, then there would be serious ramifications for this species' survival.

## Chapter 6 Spatial and temporal patterns in *Nematalosa papuensis* reproduction

### 6.1 Introduction

A diverse range of reproductive strategies are employed by tropical freshwater fish, with some following seasonal cycles (Guerrero *et al.*, 2009) and others breeding all year round, depending largely on whether they are found in the lotic or lentic habitats (Fernando, 1994).

Water temperature and photoperiod are usually the most important factors regulating reproduction in temperate species (Lam 1983); while in the tropics rainfall is often the most important cue initiating fish reproduction (Reynolds & Moore, 1982; Lamba *et al.*, 1983; Singh & Singh 1987; Machado-Allison 1992, 1994). Physicochemical changes in water chemistry and habitat restructuring as a consequence of the onset of wet season in floodplain rivers can also have effects on reproductive functions (Machado-Allison, 1987). Conditions that alter habitat structure may control measures of fish reproductive activity such as fecundity, egg size, gonadosomatic index (GSI), size at sexual maturity, and length of spawning period. In addition, the success, or otherwise, of reproductive strategies can be influenced by alteration and/or degradation of the quality of their habitat by pollutants resulting from human activities (Lloyd, 1992; Jobling, 1995; Lawrence & Hemingway, 2003; Newman & Unger, 2003).

Clupeids employ a variety of reproductive strategies. However, despite the prominence of this group in world capture fishery production many of these strategies are not well understood (Whitehead, 1985a, b; Milton *et al.*, 1996), especially in tropical fresh waters (Fernando, 1994). Whitehead (1985b) offered two explanations for the lack of understanding of the reproduction of tropical clupeids: firstly, their commercial potential may not have been as fully explored as that of their cold water counterparts; and secondly, in most cases warm water fisheries support subsistence harvesting for food security rather than a cash economy, so there is no commercial imperative in understanding their natural history. However, this situation is changing rapidly as tropical fish stocks are being depleted at an alarming rate as a result of growing population demands as opposed to stock replenishment (FAO, 2007; FAO, 2009).

Some clupeids are multiple spawners (Alheit, 1989), which should be advantageous for short-lived species because it enables them to maintain relatively stable population sizes in unpredictable environments (Armstrong & Shelton, 1990). This spawning strategy has been established for some tropical clupeids (e.g. *Sardinella brasiliensis*; Isaac-Nahum *et al.*, 1988). Some tropical freshwater clupeids display extensive spawning migrations, especially the anadromous species, such as *Nematalosa vlaminghi* (Chubb & Potter, 1984) and *Tenuulosa ilisha* (Raja, 1985). Others, such as *N. erebi*, do not migrate to spawn (Growth, 2004), while some clupeids display seasonal (e.g.,

Blue Sprat (*S. robustus*) (Rogers *et al*, 2003)) or year-round spawning (e.g., Spliethoff *et al*, 1983; Milton *et al*, 1995; Uneke *et al*, 2010).

Unpublished OTML data show that mature *N. papuensis* can be found all year round, indicating that it is a multiple spawned. This species is found in the river channel, off-river streams, blocked valley lakes, oxbow lakes, and temporary floodplain water bodies of this system; however, its juveniles have only ever been collected from off-river water bodies (OTML unpub. data; see Chapter 3), suggesting a preference for spawning and breeding in freshwater environments and possible preference of lacustrine environments as nursery grounds. Despite these preliminary observations, no detailed investigation has been undertaken into the reproductive biology of *N. papuensis*.

Therefore, given the apparent critical importance of off-river water bodies to reproduction, this chapter is aimed at establishing key information on the reproductive strategy of *N. papuensis*, including the gonad maturity stages, fecundity, oocyte diameter, and GSI in the different habitats and during different seasons. This information would also provide a perspective on habitat use by this species as reproductive metrics have been correlated directly with environmental quality in many studies (e.g., Kime, 1995; Arantes *et al*, 2010; Benejam *et al*, 2010).

## **6.2 Methods**

### **6.2.1 Fish collection**

Sampling sites, methods and sampling design are described in Chapters 2 and 3. A total of 1485 fish greater than 60 mm length at caudal fork (LCF) was collected and frozen for later examination of sexual characteristics in the laboratory (Table 6.1). The selection of this size cut-off was based on the fact that some fish, especially males, were found to be sexually mature between 70 and 80 mm.

**Table 6.1 Number of fish collected during each trip.**

Lake	2008			2009		
	Trip 1 (Wet)	Trip 2 (Dry)	Trip 3 (Wet)	Trip 4 (Wet)	Trip 5 (Dry)	Trip 6 (Wet)
<b>OXB01</b>	109	7	14	109	30	
<b>OXB02</b>	21	38	56	18	2	
<b>OXB06</b>	42	9	26	40	14	11
<b>BOS10</b>	49	27	38	42	91	
<b>OXB05</b>	4	45	52	51	15	33
<b>DAV01</b>	107	100	62	63	29	
<b>LEV01</b>	13	25	15	48	27	3
<b>Total</b>	<b>345</b>	<b>251</b>	<b>263</b>	<b>371</b>	<b>208</b>	<b>47</b>

## 6.2.2 Fish Processing

All fish were handled fresh from capture. Fish lengths (length to caudal fork – LCF) were measured to the nearest 1.0 mm using a measuring board. Each fish was then blotted and weighed on a calibrated digital electronic balance accurate to 0.01 g. *N.papuensis* shows considerable variation in body shape, but external morphological features do not relate to sexual differences, which is a similar observation made about the Australian *N.erebi* (Puckridge & Walker 1990). Consequently, internal examination was necessary for sex determination.

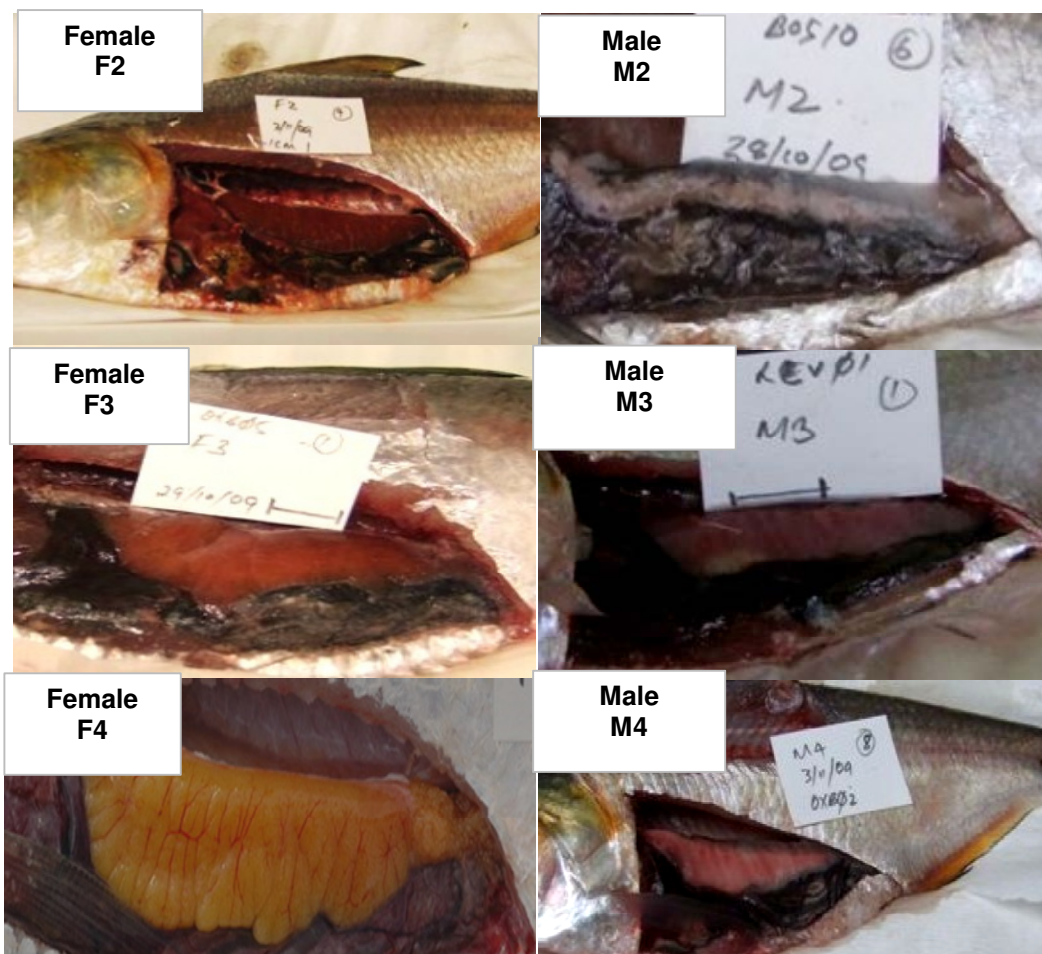
### 6.2.2.1 Maturity stages

Upon removal by dissection, the gonads were weighed (to 0.001 g), and their visible characteristics noted. All gonads were then examined under a MaggyLamp (x3 magnification) to verify sex (male, female or immature if gonads were insufficiently developed). Mature gonads (stages 3-5) were frozen and transported to James Cook University for further processing. The gonad stages for each sex were determined using the shape, colour and morphology of the gonads as described in Table 6.2 and illustrated in Plate 6.1. The gonad classification scheme used was the same as that used by OTML from 1983-1987 and employed by Hortle & Storey (2006), allowing comparison with historical data. This classification system is similar to one used by Bishop *et al* (2001) for *N.erebi* in Northern Australia.



**Table 6.2 *N. papuensis* gonad classification scale as used by OTML. Fish in resting or with sex indistinguishable were classed as stage 1.**

Stage	Description	Female Gonads		Male Gonads	
1	Immature	F1	Pink & translucent	M1	Long, thin, strap like
2	Developing	F2	Dark red & ovoid in section	M2	Milky, opaque, thicker
3	Mature	F3	Orange, indicating yolk content	M3	White and thick, trigonal in section
4	Ripe	F4	Orange, eggs can be readily separated from each other.	M4	White, thick, milt can be extruded under pressure
5	Spent	F5	Collapsed, translucent pink	M5	Collapsed, semi-opaque, milky



**Plate 6.1 Sexual stages for Male (M) and Female (F) *N. papuensis* stages 2-4.**

### 6.2.2.2 Fecundity and oocyte diameter

Gonads from 536 mature female and 471 mature male fish were frozen and transported to the laboratory. The other 478 fish were spent or were not sexually distinguishable. In the laboratory, 202 gonads were selected spanning a range of sizes from late Stage 2 to Stage 4 fish (Table 6.2). Running ripe fish (late Stage 4) were not used as eggs may have been lost during capture.

Connective tissue and any remaining fat were removed from gonads which were then weighed to 0.001 g.

**Table 6.3 Number of mature gonads (stages 2 and 4) for each season used for further analyses.**

Lake	Dry	Wet	Total
OXB01	6	7	13
OXB02	4	9	13
OXB06	15	10	25
BOS10	22	16	38
OXB05	10	17	27
DAV01	24	52	76
LEV01	6	4	10
<b>Total</b>	87	115	202

From each weighed gonad pair, one of the pair was cut in half and maturing oocytes were counted and multiplied by 4 ( for 4 halves) to give estimated number of mature eggs in a single gonad. Gonads usually would contain oocytes of different sizes classes, and for this case only gonads in the largest size class (>0.30 mm diameter) were counted, as it was assumed that these maturing vitellogenic oocytes would become ova to be spawned during the next spawning. Each sub-sample was weighed (to 0.001 g) and the number of maturing oocytes in each counted. Any maturing oocytes, which had separated from the gonad during storage and transport (i.e., prior to subsampling), were counted individually and then added to the estimated mean from subsampling. From each gonad pair, the maximum diameters of 50 randomly selected oocytes were measured under x400 magnification using an eyepiece micrometer ( $\pm 0.025$  mm) in order to calculate an average diameter of the eggs.

### **6.2.2.3 Mine impacts on Gonadosomatic Index**

McAdam *et al.* (1999) reported that GSI can be a reliable indicator of the reproductive status of fish that spawn once annually but is of limited value when fish are protracted spawners (Hails & Abdullah 1982; Rinchar & Kestemont, 1996). *N.papuensis* showed evidence of being a year-round spawner, nevertheless the logic of tracking its GSI in this system is that its GSI values year round would help in understanding any spatial variation in its reproductive health in relation to mine impacts. Additionally, the potential for any future changes in view of climate change and increased mining and petroleum activities in the headwaters of the Fly may also be assumed from this result. GSI was calculated using the following formula (Strange, 1996):

$$\text{GSI} = \left( \frac{\text{Gonad weight (g)}}{\text{Fish weight (g)}} \right) \times 100$$

#### **6.2.2.4 Data analysis**

Gonad maturity stages for each sex during the different seasons at all lakes was examined using log-linear analysis of frequency tables where frequency of occurrence was used as the dependent variable, with lake, sex, and stage the categorical predictors. The skewness of sex stage distributions was calculated to determine the direction of the distributions at each lake. The skewness would determine whether small or larger classes dominated. Fecundity, oocyte diameter, and GSI between seasons and habitats were analysed using ANOVA where fecundity, oocyte diameter, and GSI were dependent variables and the categorical predictors were lakes and season. Gonadosomatic index differences were determined separately for both sexes between seasons and lakes. The relationship between GSI and Cu was tested using simple regression analysis, using Statistica 9, with GSI as the dependent variable and Cu as the independent variable.

### **6.3 Results**

#### **6.3.1 Maturity stages**

Age at first maturity could not be estimated because of the loss of a large number of smaller size classes (<100mm). The data presented in Appendix 6.1 and Figure 6.1 show that all maturity stages were present in both sexes at each lake across a range of sizes, and throughout the year. There appears to be a peak in spawning/breeding activity during the wet season, from December to May each year as seen by the number of sexually mature fish recorded during that period (Appendix 6.1). In general, with both males and females found in either season, stages 3 and 4 were the most common. Only a small number of fish were sexually indistinguishable (N = 56 for wet season; N = 75 for dry season).

The degree of skewness of the size distribution of distinguishable sex stages is shown in Table 6.4 and Appendix 6.2, which also lists the median, mean and mode of the distribution. In general, the data reveal that fish in the mixed vegetation and grassed floodplain lakes mature later and at larger size classes, negatively skewed to the right of a normal distribution, compared to the forested lakes where fish appear to mature at smaller sizes, where mature size classes are skewed to the left of a normal distribution (Table 6.4).

**Table 6.4 Skew direction of male and female *N. papuensis*. Size of fish (median, mean and mode, in mm) are shown. Where mean>median, skew = +ve; where mean<median, skew = -ve**

Lake	Sex	Kurtosis	Median	Mean	Mode	Skew	Skew direction
OXB01	F	2.77	180	175.83	182	0.44	-
	M	0.45	152	152.86	146	-0.28	+
OXB02	F	0.62	149	152.02	163	0.78	+
	M	-0.33	130	127.11	147	-0.26	-
OXB06	F	-0.70	139.5	153.27	118	0.79	+
	M	-0.62	130	136.35	108	0.47	+
BOS10	F	-0.07	201	206.85	197	0.27	+
	M	0.37	176	174.12	180	-0.20	-
OXB05	F	1.00	229	217.98	231	-1.24	-
	M	0.02	189	184.32	175	-0.66	-
DAV01	F	0.84	215	209.87	244	-1.01	-
	M	-0.71	163.5	161.58	195	-0.09	-
LEV01	F	-0.39	197	160.87	176	-0.26	-
	M	2.70	181.5	175.07	185	-1.04	-

There are exceptions to this observation as females in OXB01, males in OXB02, and males in BOS10 were negatively skewed to the right. Fish from OXB06, a lake severely impacted by mine waste and forest dieback, had sexually distinguishable males and females positively skewed, suggesting that they mature earlier. Male and female fish from OXB05 (grassed oxbow), DAV01 (blocked valley lake), and LEV01 (mixed grass and forest fringed oxbow) all show evidence of having larger sizes (Table 6.4).

Analysis of the sex stages showed that there was no significant seasonal difference in the number of mature fish of either sex. The location of the fish, however, had a significant effect on the number of fish in each stage (Table 6.5). For both sexes, the final model included Lake (1) × Stage (3), and Season (2) × Stage (3) when the best fit was achieved (Table 6.5).

Examination of the marginal tables (Table 6.6) shows that, overall, stage 4 was the most frequently recorded maturity stage in males, occurring approximately twice as often as stages 2 and 3. In OXB01, frequency of stage 4 was consistently lower than stages 2 and 3 by an approximate factor of 33. In OXB02, frequency of occurrence of stages 2 and 3 were about the same but 3 times less than stage 4. In OXB06, the occurrence of all stages was about the same.

The frequency of stage 2 in BOS10 was half that of stage 3, while stage 4 males were 3 times more prevalent than stage 3. Stages 2 and 3 in OXB05 appeared in the population at about the same

frequency, while males of stage 4 occurred at 4 times the frequency of the other two stages. Stage 4 males in DAV01 occurred 3 times more than stage 2, and 5 times more than stage 3. This is similar to LEV01 where stage 4 males occur 3 times more than stage 3 and 6 times more than stage 2.

With the females, stage 3 was the most common, being approximately twice as frequent as stages 2 and 4. In OXB01, stage 4 females, like the males, had lower frequency than the other stages, 3 times less than stage 2 and 4 times less than stage 3. The distribution of sex stages in OXB02 was found to be the same for all. In OXB06, stage 4 females were 3 times less frequent than the other stages. In BOS10 stage 4 was found to be 10 times more frequent than stage 2, and stage 3 occurred 8 times more than stage 2. Stage 3 was the most common among females captured in OXB05, being twice as frequent as stage 2, and 3 times more frequent than stage 4. Stage 3 was the most common stage in DAV01, being 3 times more frequent than stage 2 and about 1.5 times more than stage 4. The observations in LEV01 were similar, with stage 3 two and 3 times more than stages 2 and 4 respectively.

From the results it appeared that the most fertile stages (stage 3 in females and stage 4 in males) occur more frequently in the grass floodplain lakes than in the forested reaches. In the females, the highest number of observations of stage 3 was from the grassed floodplain lakes of OXB05, DAV01, and LEV01. The high number of fish recorded in OXB01 as stage 3 females appeared to be an exception to this observation, as well as the occurrence of high numbers of stages 2 and 3 males found in this lake. Stage 4 occurrence in males of the grassed floodplain lakes (BOS10, OXB05, DAV01, and LEV01) was higher than that of forested lakes. The high occurrence of stage 4 in OXB02 may be due to the features of this lake being altered significantly by sedimentation and forest die-back to become essentially a grassed floodplain lake, with grassland riparian vegetation.

**Table 6.5 Log-linear analysis results of the frequency of maturity stages found during the different seasons and at different lakes for factors: Lake (1) x Season (2) x Stage (3).**

Sex	Factors	Maximum Likelihood $\chi^2$	df	p	Delta Likelihood $\chi^2$	Delta df	Log-linear fit $\chi^2$
<b>Male</b>	12, 13, 23	37.616	12	0.0018			
	12, 13	43.573	14	0.0007	5.957	2	0.050869
	12, 23	304.5	24	0	266.884	12	4.08x10 <sup>-50</sup>
	13, 23	136.81	18	0	99.194	6	3.7x10 <sup>-19</sup>
<b>Female</b>	12, 13, 23	48.044	12	0.0000			
	12, 13	53.495	14	0.0000	5.451	2	0.065513
	12, 23	139.32	24	0.0000	91.276	12	2.79 x10 <sup>-14</sup>
	13, 23	72.256	18	0.0000	24.212	6	0.000477

**Table 6.6 Marginal Table (Frequency + Delta) for Lake by Stage for both sexes at all lakes.**

	OXB01	OXB02	OXB06	BOS10	OXB05	DAV01	LEV01	Total
<b>Males</b>								
<b>2</b>	36	15	32	10	12	23	8	136
<b>3</b>	122	13	27	19	10	14	17	222
<b>4</b>	5	41	32	95	22	74	34	303
<b>Total</b>	163	69	91	124	44	111	59	661
<b>Females</b>								
<b>2</b>	14	22	21	7	28	27	20	139
<b>3</b>	45	18	25	55	54	96	45	338
<b>4</b>	15	20	10	67	19	77	13	221
<b>Total</b>	74	60	56	129	101	200	78	698

### 6.3.2 Fecundity and oocyte diameter

Fecundity ranged from a mean of  $22,688 \pm 5,033$  to  $126,812 \pm 14,916$  eggs per fish, and the oocyte diameters ranged from  $0.41 \pm 0.05$  mm to  $0.58 \pm 0.01$  mm (Appendix 6.2). These results show that *N.papuensis* is clearly a highly fecund species with maturity stages present in the populations all year round as shown in Appendices 4.2 and 6.1. ANOVA showed that there was a significant interaction between lakes and season (Table 6.7; Figure 6.1). There was no difference in seasons except during the dry season at DAV01, which was not different from LEV01 during the wet season. There were no significant main effects detected for tests on oocyte diameter; however, there was a significant interaction between lake and seasons (Table 6.7; Figure 6.2). The reason for this interaction was difficult to determine, and may have been a product of considerable heterogeneity of variance.

Regression analysis of  $\log_{10}$ Length against  $\log_{10}$ Fecundity shows a highly significant relationship between fish size and fecundity at the grassland lakes ( $p < 0.0001$ ; Table 6.8; Figure 6.3). For the forested lakes, there was no relationship between fish length and fecundity at OXB01 and OXB02 but the relationship was significant at OXB06 ( $p = 0.0038$ )

**Table 6.7 Summary of results of 2-Way ANOVA of fecundity and oocyte diameter in all lakes.**

Parameter tested	Effect	SS	df	MS	F	p	Tukey's HSD Test
<b>Fecundity</b>	Lake	4.496	6	0.749	6.75	0.00000	
	Season	0.167	1	0.167	1.50	0.22165	
	Lake x Season	1.672	6	0.279	2.51	0.02332	DAV01 Dry >All lakes but = LEV1 Wet
	Error	20.426	184	0.111			
<b>Oocyte diameter</b>	Lake	0.018	6	0.00296	1.984	0.06996	
	Season	0.000	1	0.00020	0.131	0.71784	
	Lake x Season	0.021	6	0.00349	2.340	0.03348	DAV01 Wet> OXB06 Wet but = all others
	Error	0.274	184	0.00149			

**Table 6.8 Linear regression relationship between length and fecundity in all lakes. ns = Not significant.**

Lake	N	df	F	p	r <sup>2</sup>	Linear equation
OXB01	13	1,11	1.558	0.237	0.046	
OXB02	13	1,11	4.762	0.052	0.402	
OXB06	21	1,19	10.865	0.0038	0.364	Log <sub>10</sub> Fec = 2.928 + 0.734 x Log <sub>10</sub> Length
BOS10	38	1,36	120.707	<0.00001	0.770	Log <sub>10</sub> Fec = -5.284 +4.270 x Log <sub>10</sub> Length
OXB05	27	1,25	257.333	<0.00001	0.911	Log <sub>10</sub> Fec = -14.536 + 8.102 x Log <sub>10</sub> Length
DAV01	76	1,74	308.907	<0.00001	0.807	Log <sub>10</sub> Fec = -10.238 + 6.366 x Log <sub>10</sub> Length
LEV01	10	1,8	151.565	<0.00001	0.95	Log <sub>10</sub> Fec = -7.748 +5.347 x Log <sub>10</sub> Length

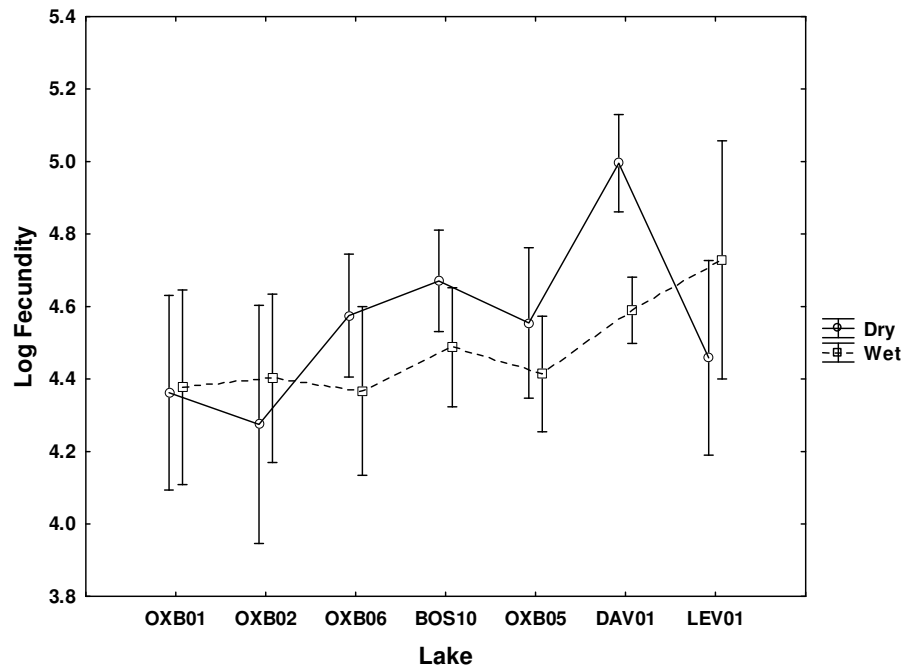


Figure 6.1 Interaction plot of mean fecundity of *N. papuensis* ( $\pm 95\%$  CI) in the different lakes and during the different seasons where  $F(6, 187) = 2.5102$ ,  $P = 0.02332$ . Lakes were arranged from upstream to downstream. Lines joining the points are to facilitate following of the seasonal patterns.

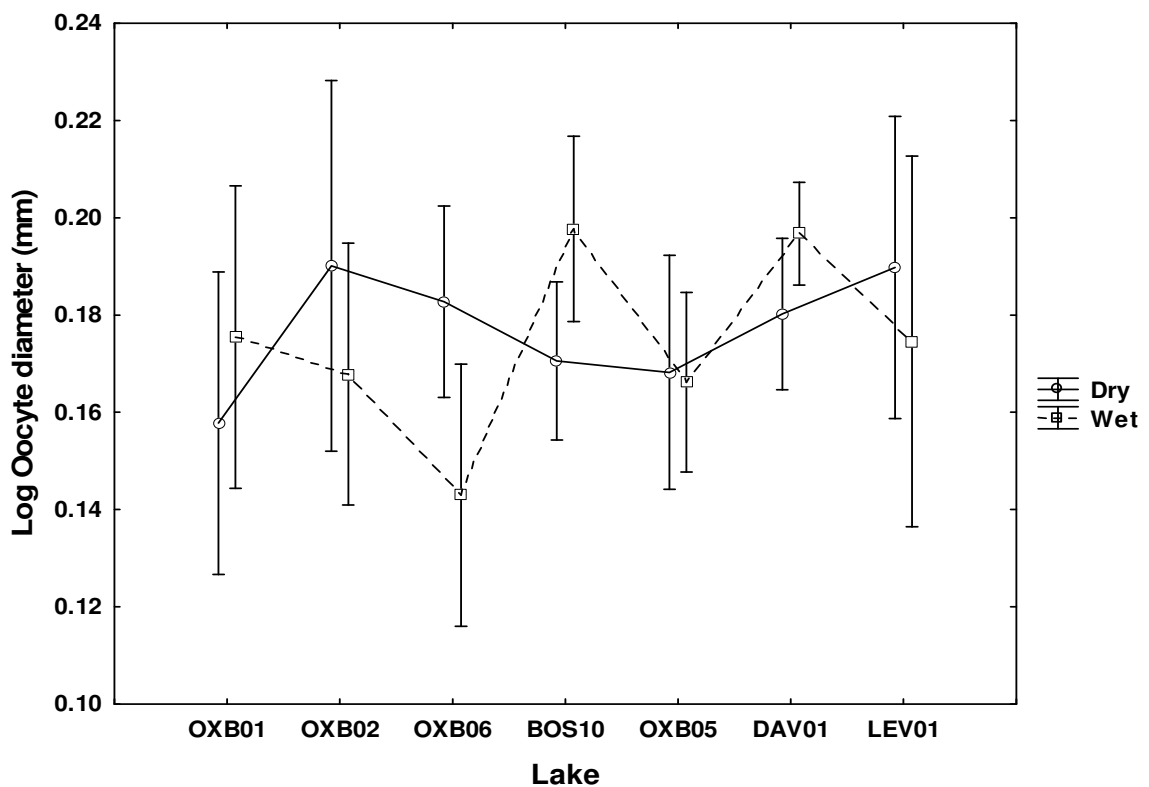
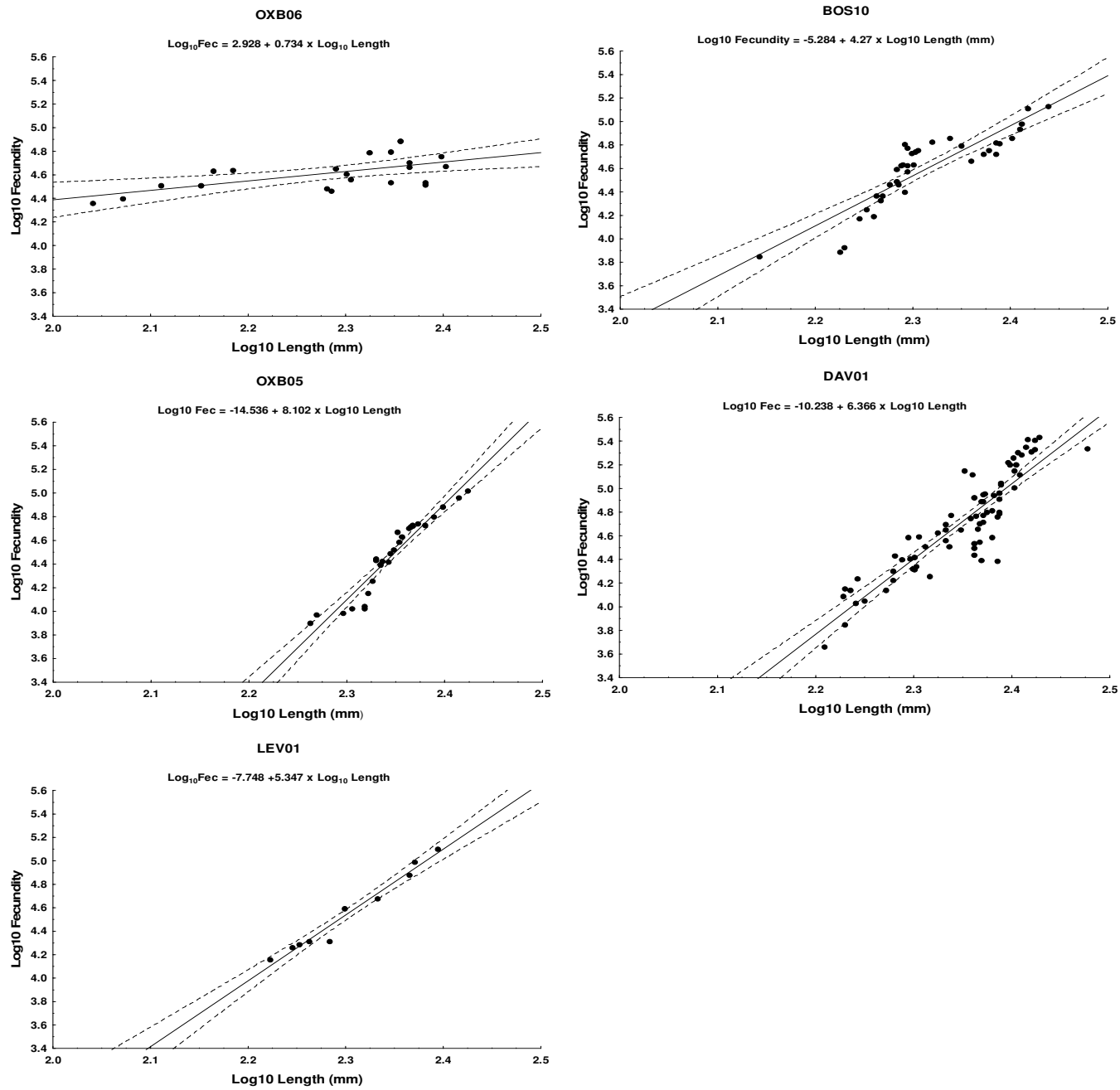


Figure 6.2 Interaction plot of mean oocyte diameter ( $\pm 95\%$  CI) in the different lakes and during the different seasons where  $F(6, 184) = 2.3400$ ,  $P = 0.03348$ . Lakes were arranged from upstream to downstream. Lines joining the points are to facilitate following of the seasonal pattern.





**Figure 6.3 Regressions, with 95% confidence intervals, between fish length and fecundity for lakes where the regression was significant**

### 6.3.3 Gonado-somatic index variation and relationship with Cu

#### 6.3.3.1 GSI variation

There was considerable variation in male and female GSI in all lakes and between seasons despite the fact that *N. papuensis* is a year-round spawned (Appendix 6.1). Female GSI in these lakes fell within a range of  $1.31 \pm 0.20$  to  $4.38 \pm 0.30$ , and male GSI was within a range of  $0.59 \pm 0.08$  to  $2.47 \pm 0.21$ . There were strong interactions detected between lakes and seasons for both males and

females (Table 6.9; Figure 6.7) indicating that lakes and seasons both have an effect on fish GSI. As would be expected there is a generally a similar trend in both males and females in all lakes when males and females get into condition at the same time for spawning during both seasons year round.

The interaction between lakes and seasons was mainly caused by seasonal differences in GSI in OXB01, OXB06, and DAV01 (Figure 6.7) and not the other lakes. The GSI of males in OXB01 and OXB06 show seasonal differences while there were no seasonal differences in males of the other lakes. The significant variation in GSI between seasons in OXB01 (Figure 6.4) may be due to resource availability, with food scarcer in the dry season than the wet as noted in Chapter 5.

**Table 6.9 Results of 2-way ANOVA comparing GSI between lakes and seasons for each sex.**

Comparison	Effect	SS	df	MS	F	p
<b>Females</b>						
	Lake	3.0555	6	0.5092	13.312	0.0000
	Season	0.1256	1	0.1256	3.283	0.0704
	Lake*Season	3.0547	6	0.5091	13.309	0.0000
	Error	26.7014	698	0.0383		
<b>Males</b>						
	Lake	2.4397	6	0.4066	13.764	0.0000
	Season	0.1504	1	0.1504	5.091	0.0244
	Lake*Season	1.5665	6	0.2611	8.838	0.0000
	Error	18.7596	635	0.0295		

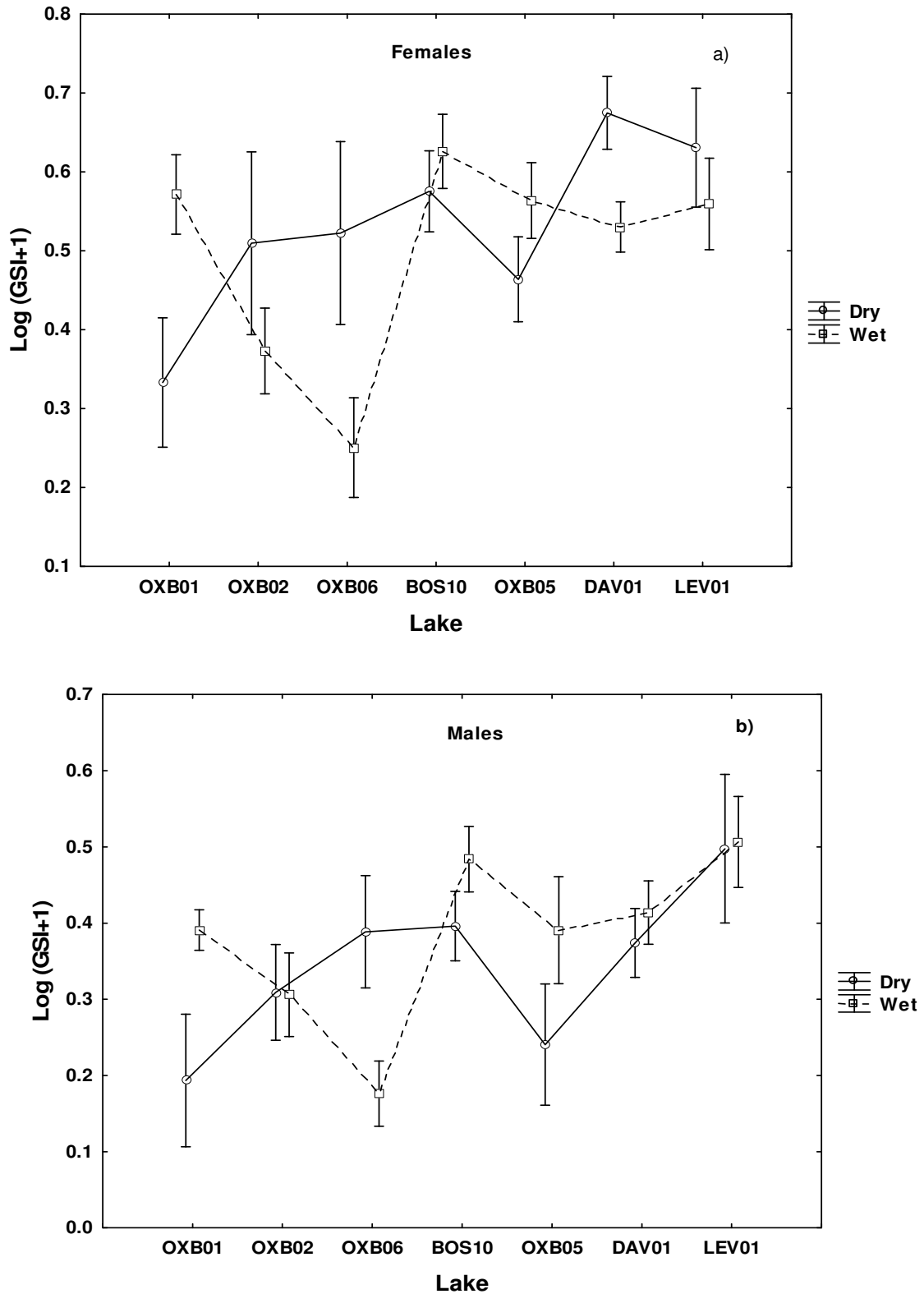


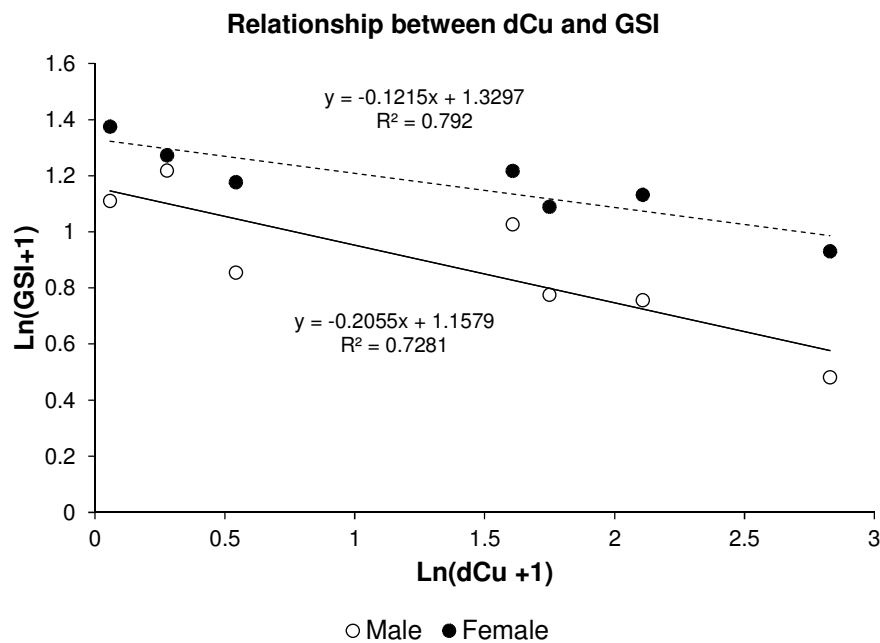
Figure 6.4 Interaction plots of Lake x Season for a) females only:  $F(6, 698) = 13.309, P < 0.00001$ , and b) males only:  $F(6, 635) = 8.8376, P < 0.00001$ . Lakes were arranged from upstream to downstream. Lines joining the points are to facilitate following of the seasonal patterns.

### 6.3.3.2 GSI relationship with Cu

There is strong evidence of higher levels of Cu in the mine-impacted lakes affecting the reproductive ability of fish (Table 6.10; Figure 6.5). The results suggest that with increase in dCu in the water column there is a corresponding decrease in GSI. This relationship was observed in both male and female fish. It can be noted in section 6.3.3.1 (Figures 6.4) that OXB06, the ORWB with the highest levels of dCu, had a significantly lower GSI in the wet season compared to the dry season. This may be a response to greater flushing of pollutants into this lake during the wet season

**Table 6.10 Results of regression analysis of GSI against dissolved Cu.**

Sex	N	dF	F	p	r <sup>2</sup>	Linear equation
Female GSI	7	1,5	19.0441	0.007	0.792	Y= -0.122X + 1.33
Male GSI	7	1,5	13.3885	0.015	0.728	Y= -0.206X + 1.158



**Figure 6.5 Regression plots of *N. papuensis* GSI and mean dissolved Cu in the lakes.**

## 6.4 Discussion

### 6.4.1 Maturity stages

A wide variety of reproductive strategies is employed by species in the tropics with some governed by seasonality (Guerrero *et al*, 2009), and others breeding throughout the year, depending largely on whether they are found in the lotic or lentic habitats (Fernando, 1994). Gonad maturity and spawning in *N. papuensis* is consistent with the findings that many clupeids, including some tropical species, are multiple spawners (Alheit, 1989). This study shows clearly that that mature and ripe *N. papuensis* were present in samples spread through the year, an indication that this species breeds throughout the year, but with a peak in breeding during the wet season (December-April). Bishop *et al* (2001) postulated that *N. erebi* in northern Australia also spawned through the year, with peak spawning periods during the wet season, in contrast to *N. erebi* from the temperate Murray-Darling system whose spawning is more seasonal (Puckridge & Walker, 1990).

This result is consistent with studies on similar systems elsewhere, which show that many tropical freshwater clupeid species spawn all year round (Alheit, 1989, Sirimongkontharworn & Fernando, 1994, Fernando, 1994), but are especially more reproductively active during the wet season when resources such as food and habitat become abundant (Bye, 1989; Silvano *et al*, 2003). Multiple spawning should be advantageous for short-lived species like *N. papuensis* because it is a strategy for them to maintain relatively stable population sizes in unpredictable environments (Armstrong & Shelton, 1990; Fernando, 1994).

The results also show that in the grassed and mixed vegetation lakes the more reproductive males and females are found in the larger size classes, with a size class distribution negatively skewed, to the right of a normal distribution chart. The opposite is the case with mature males and females in the forested lakes except for females in OXB01 and males in OXB02. Most of the fish in these size classes were found in the mixed and grass vegetated floodplain lakes indicating that these habitat types were more favourable for fish reproduction in this floodplain river system. The results of Chapter 5 of this thesis show that there is a greater abundance of their preferred food items in the mixed vegetation and grass floodplain lakes of this system, especially during the wet season, thereby providing more resource for the fish in these habitats to be conditioned for reproduction throughout the year. As *N. papuensis* is a dominant planktivore, it is expected that the greater abundance of phytoplankton in the grassed floodplain than the forested lakes (WRM, 2007) will lead to faster growth and better reproductive fitness than in forested lakes. Studies in tropical water bodies elsewhere established positive correlation between phytoplankton biomass and increase in reproductive activity (Admassu, 1996; Pena-Mendoza *et al*, 2005; Gómez-Márquez *et al*, 2008).

#### 6.4.2 Fecundity and oocyte diameter

This study found that the smallest mature (i.e. stage 3) fish found were males of 70 mm LCF and females of 103mm LCF, which are smaller than the size at first maturity found for Australian fish (Table 6.11). The ageing data discussed in Chapter 4 indicates that *N. papuensis* possibly mature in their first or second year of life and are relatively early maturers and continue to spawn through the year.

**Table 6.11 Comparison of smallest size at maturity of *N. papuensis* (this study) with those found for *N. erebi* in Australia (Alligator Rivers and Lower Murray River).**

Study	Length (LCF mm)	
	Female	Male
Fly River system, this study	103	70
Alligator Rivers Bishop <i>et al.</i> (2001)	140	130
Lower Murray River Puckridge & Walker (1990)	180	144

In tropical Northern Australia, Bishop *et al* (2001) reported a *N. erebi* fecundity range of 80 000 to 230 000 (mean = 132 000) from three mature females and mean oocyte diameters in two ovaries of between  $0.43 \pm 0.04$  mm and  $0.41 \pm 0.05$  mm' confirming the observation that *N. erebi* is a very highly fecund species with and very small eggs (Briggs 1980), which is similar to *N. papuensis* in the Fly River system. Similarly, in the Murray-Darling River system, *N. erebi* fecundity was also high, with 33 000 to 880 000 eggs for fish of 199 mm TL (175.12 mm LCF) and 403 mm TL (364.64 mm LCF) respectively (Puckridge & Walker, 1990).

Therefore, while *N. papuensis* in the Fly has similar fecundity to *N. erebi* in tropical Australia, both have fecundities that are possibly lower than that determined for the temperate Murray River system (Puckridge & Walker, 1990). It is of some interest to compare the Fly system fish with those from the Murray River. *N. papuensis* from the Fly (this study) have generally half the number of eggs of similar-sized *N. erebi* in the Murray River (noting that the sample size was very low for the Murray). Murray River fish spawn only once per year, so this difference may be explained by Fly system fish spawning more than once each year, with fewer eggs produced each spawning, but with perhaps greater total egg production each year. These strategies may be environmentally induced and dependent on the latitude where the species is found. Since food is available in the Fly system all year round multiple spawning would be supported but it would peak during the wet season. This was observed for the clupeid *Limnothrissa miodon* in African lakes

(Spliethoff *et al*, 1983). In marine multiple spawning clupeids of Solomon Islands fecundity was related to food availability and location (Milton *et al*, 1995).

*Nematalosa papuensis* is clearly a highly fecund fish whose population size is unlikely to be limited by the number of eggs produced each year, as even a few thousand fish can spawn many millions of eggs several times each year. The recruitment to adult size of fast-growing fecund spawners such as *N. papuensis* is typically limited by competition among larval or juvenile fish for food (e.g. Okamoto *et al*, 2014) so that removal of a large proportion of the adult fish has little effect.

This study indicated that fish in different lakes had different fecundity, and there was seasonal influence on fecundity, depending on the location of the lakes. This may be related to water quality issues at each lake being influenced by seasons. There was a general increasing trend in fecundity downstream from forested into the grass floodplain lakes, probably indicative of better habitat and food availability with increasing distance from the forested lakes and with increasing distance from the mine. Drawing from the results of Chapter 4, where fish size in the grass and mixed vegetation lakes were shown to be significantly higher than those from the forested lakes, it is not unexpected that fecundity would be higher in the grassed floodplain and mixed vegetation lakes since larger fish are more fecund. This trend was present during both the wet and dry seasons.

There was only a weak significant interaction between lakes and seasons for egg diameter, largely because of the oocyte diameter at OXB06 being lower than those found in fish at BOS10 and DAV01. In general, though, oocyte diameter was not influenced by habitat or seasons, indicating that this species maintains its reproductive fitness by maintaining its oocyte sizes across all habitats, seasons, and sizes. Puckridge & Walker (1990) similarly found that it is likely that egg size does not change significantly with body size in *N. erebi*. In some fish species like the Pacific Salmon, egg size is correlated with female body size (Fleming & Gross, 1990; Beacham & Murray, 1993). In other species like pipefishes, egg size correlated with female body size in those with pouches, but was similar across female sizes in the species lacking pouches (e.g. Goncalves *et al*, 2011). Yet in other species reproductive strategies employed are size-independent fecundity and egg-size, where fewer larger eggs are more for quality offspring compared to shortlived pelagic species which spawn thousands of small eggs to maximize survival numbers (Duarte & Alcaraz, 1989).

#### **6.4.3 Effect of pollution on reproduction**

There is evidence from this study that suggests that the low GSI observed in lakes most severely impacted by mine wastes during the wet season, OXB02 and OXB06 are a consequence of the effect of pollutants. Schulz & Martins-Junior (2001) found that *Astyanax fasciatus* located downstream of a heavy metal pollution source had significantly lower GSI and egg diameter than

fish found in un-impacted reaches. Jobling *et al* (2002) found that wild roach located in areas severely impacted by treated sewerage had severe gamete alteration and altered sexual maturation. In addition, it has been reported that *Cyprinus carpio* located in waters severely impacted by heavy metals had significantly lower condition and fecundity than fish from un-impacted waters (Benejam *et al*, 2010). The lowered GSI during the wet season may be an adaptation to the regular heavy flushing of sediments and metals into these water bodies during the wet season, whilst the GSI during the dry seasons were significantly higher in comparison.

It can also be assumed that fish in the Fly River system mature in their first or second year, based on cohort analysis and gonad stage to length relationships (Chapter 4; Hortle & Storey, 2006). In temperate Australia, *N. erebi*, mature during their second year, whereas with faster growth rates in the tropics some fish mature in their first year. In ORWBs such as OXB06, because of water quality issues, *N. papuensis* may actually resort to building up fat storage during the wet season, to avoid pollution issues and to spawn during the dry season when there is lower flushing of metals and other pollutants into the system. Such a strategy, building up fat reserves for reproduction later has been reported for other species (e.g. Esteves & Pinto Lobo, 2001; Torres-Mejia & Ramírez-Pinilla, 2008; Ballesteros *et al*, 2009).

## 6.5 Conclusion

Apart from some brief work undertaken on the reproductive biology of this species by Hortle & Storey (2006), no other detailed study has been undertaken on primary consumers such as *N. papuensis* in the mine-impacted Fly River system. It is assumed that any work on primary consumers would clearly reveal the second level of impact of pollutants before there is a noticeable community-wide consequence of habitat degradation. This study has confirmed observations by Hortle & Storey (2006) that *N. papuensis* is a year-round spawned, with fish from the grassed floodplain lakes more fecund than those from other habitats. Evidence from this study also revealed that there is a possible linkage between resource availability and fecundity or reproductive fitness. A clear revelation from this work is that habitat degradation through pollution by mining waste, especially Cu, is negatively affecting the reproductive ability of this species. These results, therefore, imply that not only is the reproductive biology of this species, but that of other primary consumers and possibly some top order feeders have now been impacted by the degraded habitats along the mine-impacted corridor of the river system.

With the looming impacts of climate change coupled together with mine impacts, populations of species in the polluted Fly River lakes and river channel may be under serious threat. The domino effect on higher-order consumers will be declining biomass across the system as probably is already the case where massive declines have been reported by various OTML reports. This work therefore reveals the need for a concerted effort by the government of Papua New Guinea and the



mining, and oil and gas companies operating in the head waters to establish more environmentally sound waste management and mitigation strategies and conservation of wetlands for future recolonisation of the Fly River lakes.

## **Chapter 7 General discussion: biology of *Nematalosa papuensis* in a mine-impacted environment and its potential for a future fishery**

This chapter summarises the findings of this research on the biology of *N.papuensis* in the context of its importance as a keystone fishery species, and the implications of current and future predicted environmental changes in its habitat, and the implications on its biology.

### **7.1 Habitat preference**

This study was undertaken, as stated elsewhere, along the impacted stretch of the Fly River covering forested, grassed floodplain lakes and lakes that are of mixed forest and grass fringed vegetation. An important result of this study was the finding that a key driver in *N. papuensis* biology is habitat type.

This species is more abundant in the open grassed floodplain lakes than other habitats, implying that the former habitat is more productive than the forested lakes, whilst the river channel has always yielded very low catches of this species as reported in numerous OTML reports (e.g. OTML, 1995). Numerous freshwater clupeids elsewhere have also been reported to have a preference for open pelagic waters where they feed as planktivores, and breed year round (e.g. Lazzaro, 1987; Munyandorero, 2002).

This work has also established that most of the natural habitat of *N. papuensis* along the Fly River corridor has now been negatively impacted by mine derived pollution, posing a serious threat to aquatic and riparian flora and fauna. Any alteration of habitats, therefore, will leave behind a chain reaction of biological adjustments in response to these modifications for any species. Ultimately the level of impact and the resulting quality of the habitat will determine the survival of the species. In some areas, as seen in this study, the *N. papuensis* habitat has already been polluted by the intrusion of excessive sedimentation and metals in the ecosystem.

Although this study is not unique in the sense that others have documented the decline in water quality in the Fly River corridor lakes, it is unusual in that this is the first time for any study to focus on the comparison of different lake habitats of this floodplain system. This study added some new details on vegetation and water quality with the primary purpose of establishing an understanding of the habitat in which *N. papuensis* dominates. The results clearly reveal the fact that much is still unknown about large tropical freshwater lake habitats and the ecological role they play in this region of the world.

In light of the findings of this work, any proposed fishery development based on this species would need to consider the potential yields of the grassland and forested lakes separately with a view to establishing separate fishery management plans for the different habitats.

## 7.2 Population dynamics and structure

This study of the population dynamics and structure of *N. papuensis* reveal that its populations are quite variable in space and time, and have consistently declined in the lakes downstream of the Ok Tedi mine over time. Activities such as mining, agriculture, shipping, tourism, industrial development, waste dumping, tourism, and dam construction have been identified as impacting the marine and coastal environment and therefore fisheries globally (Hongkul, 1999; Newman & Unger, 2003).

The effects of mining activities on fish are well documented globally. In the United States, Daniel *et al* (2015) reported that even a single mine upstream of a river system can cause massive changes by damaging fish habitats and therefore fisheries downstream. This is because water quality can be affected by changes in nutrient composition; changes in the natural physicochemical properties of the water body, or through the invasion of pristine environments by pollutants associated with anthropogenic activities (see for example, Kadlec, 2003; Thomas *et al*, 2003; Kantawanichkul & Neamkam, 2003). When fish and other organisms experience environmental disturbances which lie outside the normal range, the effects can be dramatic and long-term exposure to less severe disturbances may result in severe consequences to individuals in populations (Jobling, 1995).

In Africa, fisheries and marine resources in Nigeria were reported to be facing total collapse or extinction, due to over-fishing and destruction of marine life and natural habitats by the pollution of water bodies. Furthermore, the unregulated and excessive use of pesticides for fishing and the deliberate disposal and dumping of toxic and hazardous wastes into water bodies have caused massive fish kills and loss of aquatic life and habitats (Adeyamo, 2003). In many cases the impacts of pollution at the population and community level are a manifestation of pressures acting together rather than a single pollutant being responsible (Elliott *et al*, 2003).

### 7.2.1 Changes in the Fly River lakes

Long-term declining trends in the biomass of *N. papuensis* in mine-impacted lakes of the Fly River, shown in this study, reaffirmed various reports of declining fish catches in mine impacted reaches of this river system (referenced previously). The inferences from the biomass trends in the mine-impacted lakes are that, first, fish ecology and fisheries of the Fly River system are under severe threat from current mine practices of direct dumping of tailings and waste rock into the river system. This suggests that mine impacts are a causative factor in the reduction of *N. papuensis* populations in the long term, as opposed to short-term *El Nino* effects from which populations could recover rapidly. Secondly, the results imply that any hopes for the recovery of fisheries in the mine-impacted ORWBs depend on establishment of conservation management strategies for lakes with viable populations away from mine-impacted reaches and lakes of the Fly

River. This is so that with the termination of mining activities, populations from these other water bodies maybe used to recolonise the impacted lakes. The recovery of the Fly River ecosystem from the environmental damage may actually never happen within a lifetime because it has been predicted that the negative impacts will continue many decades after mine closure (Pickup, 2009; Pickup & Cui, 2009).

### **7.2.2 Changes manifested in populations**

The ultimate effect of contaminants in the aquatic system is usually manifested visibly in changes at the population and community levels. Human influence is central to the changes observed in the environment. More often, effects to individuals are measured with the intent to predict consequences to interaction among different species and in populations. Breitburg *et al* (1997) pointed out that the degree of variation among interacting species in their response to negative changes to the physical environment is likely to be important in determining when species or population-level processes will strongly affect attributes measured at higher levels of ecological organisation.

It has been reported that long-term changes in genetic qualities may also occur over a prolonged exposure period, altering population dynamics. For example, the rodents *Microtus arvalis* and *M. rossiaemeridionalis* living near the Chernobyl nuclear reactor showed large changes in DNA structure over time (Baker *et al*, 1996). In fish, Murdoch & Hebert (1994) found that there was reduced variability in mitochondrial DNA of the brown bullhead (*Ameiurus nebulosus*) from the Great Lakes which was attributed to pollutant-induced reductions in effective population size. Kopp *et al* (1992) also demonstrated that there was significant reduction in genetic heterozygosity in stressed populations of the central mud minnow (*Umbra limi*) in a contaminated environment, which indicated pollutant-related loss of genetic diversity. Such observations in a polluted environment may also be attributed to selection (Battaglia *et al*, 1980; Gillespie & Guttman, 1989; Newman *et al*, 1989; Heagler *et al* 1993; Keklak *et al* 1994; Schlueter *et al* 1995). Fish populations under long-term exposure to sublethal doses of toxicants have been reported to have individuals with reduced sizes, altered sexual maturity times, reduction in spawning periods, and decreased lifespan (Moiseenko, 2002)

The main consequence of prolonged exposure by aquatic fauna to contaminants was that much of their energy was spent on producing defense proteins (such as cytochrome P-450) at the expense of growth and reproduction (Newman & Unger, 2003) therefore impacting on the next generation. Sibly & Calow (1989), Holloway *et al* (1990) and Sibly (1996), describing the energy allocation of individuals to different tasks, noted that these responses can produce significant changes in population demographics.

The general impacts of pollutants such as sedimentation and chemicals, particularly heavy metals, has been postulated to be the major causes of decline in fish populations currently experienced in the Fly River system in Papua New Guinea (Smith *et al*, 1990; Swales *et al*, 1998a; Swales *et al*, 2000)

### **7.2.3 Changes manifested in community response**

The community is made up of species that interact and form an organized unit (Magurran, 1988). A number of studies on the adverse effects of pollutants on predator-prey interactions have shown that there is the likelihood of pollutants enhancing the extinction of a species population, even if the toxicant concentrations are below those causing diminished growth, reproduction and individual survival fitness in the population (Moiseenko, 2002). Various studies have shown that fish population and community survival are impacted by metals (Kania & O'Hara, 1974; Sullivan *et al*, 1978); natural factors such as  $\gamma$ -irradiation effects (Goodyear, 1972) or weather patterns and nutrient flushing (Swales *et al*, 1998a).

Sedimentation and heavy metal impacts have been shown to be the responsible agents of change in the community and population structures of fish in the Fly wetland system over 20 years. This was clearly evident in the gradual shifting of fish community structure in mine-impacted areas as opposed to non-impacted areas (Smith *et al*, 1990; Smith & Morris, 1992; Swales *et al*, 1998 a, b). Swales *et al* (1998b) reported that the analysis of temporal and spatial changes in community structure due to the impacts of the Ok Tedi mine on the aquatic environment identified several consistent patterns over time. They reported that the analyses of other Fly riverine sites showed a consistent pattern of gradual changes in community structure with time at each site. These changes were greatest (i.e. total spatial separation of time periods in ordination space) at sites closest to the mine, with increasing overlap (i.e. less separation/reduced change in community structure; *viz.* less impact) between time periods with increasing distance from the mine.

Such impacts are also seen in other taxonomic groups. Chadwick and Canton (1983) for instance showed that functional guilds or groups among macroinvertebrate communities can be modified by toxicants. Other workers (Niederlehner *et al*, 1985; Cairns *et al*, 1986, McCormick *et al* 1991) have also demonstrated a clear toxicity concentration response relationship for colonisation by protozoa of artificial substrates. Roussel *et al* (2007) showed that long-term exposure of phytoplankton, periphyton and macrophyte communities resulted in significant effects on the structure of their communities. This would of course affect fish populations because of the destruction of algal and the planktonic communities which are the primary producers.

Short-term, large-scale community structure changes are usually noticed when high nutrient fluxes cause eutrophication and therefore high biological oxygen demands in water bodies – for example, as a result of *El Niño* events (Swales *et al*, 1998a, b). Hypoxia can change fish assemblage composition and may lead to mortality if fish cannot escape to more oxygenated waters (Smale & Rabeni, 1995 cited in Killgore & Hoover, 2001). For example, species richness, abundance, and size of fish were substantially reduced at hypoxic levels, suggesting a threshold response level of assemblage composition (Jobling, 1995; Killgore & Hoover, 2001).

The depletion or disappearance of sensitive species within a community may provide an early warning of deteriorating conditions and therefore implications for a community. An example is the disappearance of a pH-sensitive species, the mysid shrimp *Mysis relicta* from an experimental lake that was undergoing acidification (Schindler, 1996). This species provided an early warning of the worsening physicochemical conditions in this Canadian lake.

#### **7.2.4 Implications of climate and environmental changes on population dynamics**

The observed changes in fish assemblages of the Fly River system were evidently in response to the altered physical habitat of the ecosystem as reported by various authors (e.g. Swales *et al*, 1998; Storey *et al*, 2008; Storey *et al*, 2009b). These changes are highly likely to be exacerbated when more sedimentation due to increased flooding, increased metal levels as well as a whole suite of changes that may occur due to climate change can be speculated from the observations as made by Swales *et al* (1998) and in this study.

It is inevitable that the Fly River system will face *El Niño* and *La Niña* events in the future as part of the normal short term climatic variations. When the former occurs, then the events of the *El Niño* years as seen in the Fly River in the past will repeat themselves, where massive fish kills happen during the drought years, as reported by various authors. On the other hand, with an increase in rainfall during the *La Niña* events the efflux of pollutants into the of river water bodies will also increase. The impacts of both events associated with mine and climate change scenarios are as summarised in Figure 7.1.

#### **7.3 Diet**

*N. papuensis* is primarily a phytodetritivore, confirming results of other studies, contributing up to approximately 40 % of the biomass of the total fish biomass of the Fly River system, making it stand out as the single most important contributor to carbon recycling in this ecosystem. This level of abundance of a primary consumer clearly indicated that there is high plankton productivity and biomass in the floodplain lakes of this floodplain system. Similar observations have been reported in other regions. In America, for example, Allen *et al* (2000) reported that the gizzard shad and threadfin shad density and biomass were positively related to levels of chlorophyll 'a'. In Lake

Tanzanyika, Kimirei & Mgaya (2007) noted a positive correlation between the levels of chlorophyll 'a' and clupeid catches. On a global scale this relationship was shown when primary production was drastically reduced, and therefore planktonic food sources, resulting in a corresponding reduction in the Chilean and Peruvian fishery, which is based on herrings, during the *El Niño* years of 1997/1998 (FAO, 2007).

In the Fly River system, habitat degradation in mine-impacted lakes has not only caused long-term decline in fish catches and diversity (Swales *et al*, 1998; Storey *et al*, 2009b), but has also caused corresponding changes to plankton assemblages (WRM, 2007; Stauber *et al*, 2009). With a decline in its food supply, it is expected that many other biological functions of *N. papuensis*, such as reproduction, predator avoidance and migration, will be negatively affected leading to changes in population size and structure.

Various workers have shown that increases in bioavailability of metallic ions in aquatic systems will result in changes in primary producer assemblages (Stauber *et al*, 2009) and in their contaminant uptake (Pflugmacher *et al*, 1999; Sanchez-Rodriguez *et al*, 2001; Bowles *et al*, 2001), leading to their biomagnified availability in the diet of higher consumers (e.g., Sandermann, 1992; Bowles, *et al*, 2001; Nystron *et al*, 2002). Chronic exposure of phytoplankton, periphyton, and macrophyte communities to dissolved metals has been reported to have had significant negative effects on the structure of their communities (Roussel *et al*, 2007). Apart from the smothering of productive habitat by sedimentation, it is likely that mine-derived dissolved metals such as Cu are binding with sulphate ions (under the right conditions) thereby becoming one of the many mechanisms involved in metal toxicity to algae. Sulphate ions are commonly available in hypolimnetic waters of tropical lakes (e.g. Payne, 1986) to form copper sulphate, which is a common algaecide (e.g. Chen & Lin, 2001) and could therefore be available to periodically reduce phytoplankton in the Fly River water bodies. These reports lend support to a core result of this study, unreported elsewhere, that there is much less phytoplankton in the diet of *N. papuensis* in lakes nearest to the mine as opposed to grassed floodplain lakes which are located further downstream.

### **7.3.1 Implications of climate and environmental changes on diet**

Climate models of the tropical Pacific suggest that most rivers and streams in the Fly region will receive increased run-off from increased rainfall of between 5-20% by 2035 and 10-20 % by 2100 (Lough *et al*, 2011; Gehrke *et al*, 2012) as well as being subjected to increased temperatures . The increase in precipitation would result in increased river discharges, with the Fly River predicted to have a 9% increase by 2050 (Palmer *et al*, 2008; Lough *et al*, 2011).

These scenarios may have a number of effects on the fisheries ecology of the Fly River system since most freshwater species have specialised habitat and dietary requirements (Ficke & Myrick, 2004). Firstly, there will be more new freshwater habitats, increased productivity, increased floodplain connectivity, and increased nutrient delivery into lakes. A consequence of this setting would be that there will be an increase in flooding, erosion and sedimentation downstream. However, as a large part of the sediment would be mine-derived and laden with heavy metals, the current mine-derived impacts may be exacerbated. Given the potential for both positive and negative effects of climate change, further investigation are required to determine whether the modelled scenarios will be positive or negative for the Fly River and its fisheries ecology, especially in the mine-impacted regions.

In addition, phytoplankton community structure, for example, can also be sensitive to climate change, resulting in floristic structural shifts in relation to mixed layer depth, shoaling and increased stratification in response to increased temperatures (Karl *et al.*, 2001). Changes in community structure as a consequence of increased temperatures could therefore profoundly affect the capacity of the aquatic biological community to draw down atmospheric CO<sub>2</sub> for transportation into the deep waters (Arrigo *et al.*, 1999). Modifications, therefore, in habitat, and the abundance and structure of primary producers as a consequence of mine impacts or climate change, or both, will determine the structure of primary consumers in response to availability of resources.

#### **7.4 Growth and reproduction**

Evidence from water quality data (Chapter 2) and reproductive biology (Chapter 6) suggests that *N. papuensis* may be resorting to complex behaviour to continue breeding in mine-impacted lakes in response to some form of stress, possibly triggered by environmental changes in its habitat. Results of this study indicated that the reproductive ability of *N. papuensis* is influenced by habitat type, being lower in forested lakes as opposed to open grassed floodplain lakes (Chapter 4). This observation, however, may also be an artifact of water quality issues currently impinging on the mine-impacted lakes of the Fly River. Data from this work has also suggested that reproductive fitness is better in lakes not affected by mine effluents (high dCu, and high TSS) where gonadosomatic index (GSI) for both male (MGSI) and female fish (FGSI) in lakes with little or no impacts from the mine were higher than that of fish in mine impacted lakes. Studies elsewhere have shown that prolonged exposure to contaminants have resulted in reduced sizes, decreased lifespan, reduced spawning periods, and altered sexual maturity times in fish populations (Moseenko, 2002).



#### **7.4.1 Implications of pollutants on growth and reproduction**

Pollutant concentrations, be they Cu or high TSS loads, above ambient levels, affect growth and reproduction of many fish species in aquatic systems (e.g. Swales *et al*, 2000; Shailaja & De' Silva, 2003; Styrdom *et al*, 2006; Deane & Woo, 2006)

It is known that at the sublethal levels, fish and other aquatic organisms are faced with effects such as endocrine disruption, protein and membrane damage in gonads, spermatotoxic effects, altered sex ratios, intersex, life cycle strategies, reduced recruitment and reproductive behaviour when in contact with xenobiotics (Moiseenko, 2002; Goksoyr *et al*, 2003, Griffin, 2009).

Several studies have clearly shown that pollutants have a role in negative sub-lethal effects on embryonic and larval development in fish (e.g. Perry *et al*, 1991; Stouthart *et al*, 1998). This happens because the eggs and larvae of many fish species are planktonic and will come into contact with xenobiotics in the water column or the surface micro layer (Hardy *et al*, 1987). Fish larvae may be exposed to xenobiotics while still in the egg, through water after hatching, from yolk, or from food consumed during larval development (Hylland *et al*, 2003). These sublethal impacts have been well reviewed by several workers (see reviews by von Westernhagen *et al*, 1988; Longwell *et al*, 1992; Dethlefsen *et al*, 1996).

In the case of the Fly River lakes, there is a clear relationship between the level of Cu in the system and the reproductive ability of *N. papuensis*, with elevated levels of dissolved Cu in the lakes having had a significant negative impact on the reproductive ability of the species.

#### **7.4.2 Implications of climate change on growth and reproduction**

In addition to pollutant impacts, fish in the Fly River system may face negative reproductive and growth issues associated with higher temperatures resulting from climate-change. Higher water temperatures are known to have strong effects on animal activities such as respiration, feeding, growth, reproduction, and the uptake and elimination process of toxicants such as metals (Baykan *et al*, 2006). Temperature increases will also have a profound effect on fish larval physiology primarily through changes in the rates of enzymatic reactions (Hochachka & Somero, 2002). Some changes in temperature and dissolved oxygen content of water will affect the resistance of fish to toxic chemicals or diseases (e.g. Bucke, 1993); also changes in temperature, pH, water hardness, and humic acids can affect the toxic state of some chemicals or compete with their uptake by fish (e.g. Lloyd, 1992).

The possibility of increase in hypoxic conditions, and therefore acidity, in the Fly River lakes with the current levels of sedimentation remain, especially when there are prolonged dry periods. Hypoxia has been linked to the retardation of fish development and lengthening of incubation time of early egg stages of some fish (Rosenthal & Alderdice, 1976; Giorgi & Congleton, 1984;

Keckeis *et al*, 1996). Apart from low DO, hostile environmental conditions such as physical disturbances and variable salinity can also cause certain larval aberrations such poor quality of eggs, low food reserves in the embryo, and inability to avoid predation (see Wiegand *et al*, 1989; Purceli *et al*, 1990).

Since the impacts of increase in sedimentation and dissolved metals in mine impacted lakes may increase with the effects of climate change, this will increase the threat of reproductive failure and therefore the growth of fish in the Fly River especially since it has been predicted that the impacts of the mine on the Fly River system would last for decades after mine closure. (Pickup & Cui, 2009) reported models, based on a 2010 mine closure plan, that most of the sediments now in the Lower Ok Tedi system will have been transported into the Middle Fly region by 2060, however there is uncertainty as to when this load in the Middle Fly reaches would be cleared. They have predicted that the water levels in the Middle Fly would increase thereby increasing the number of aquatic habitats with major changes to the Fly River system expected (Pickup, 2009). This then will have a domino effect on the viability of a reproducing and vibrant population especially in these lakes. It is therefore important that research is undertaken to understand clearly the combined effects of climate change and mining in the Fly River region.

## **7.5 *Nematalosa papuensis* fishery**

### **7.5.1 Regional setting**

The Indo-West Pacific region has the greatest diversity of clupeoids, with about 160 species. This is half the total number of clupeoid species known, but only providing a yield that is 7% of the world clupeoid catches (e.g. Whitehead, 1985). Whitehead (1985) offered two explanations for this apparent discrepancy: firstly, their commercial potential has not been as fully explored as their cold-water counterparts; and secondly, in most cases warm-water fisheries support subsistence harvesting for food security rather than a cash economy.

Despite this, there is a notable paucity in catch information of different species harvested in the clupeoid fishery in the tropics (Munyandorero, 2002). This deficiency in relevant information on tropical clupeoid species is surprising given that they occupy such an important position at the base of the food chain of many aquatic ecosystems (e.g. Whitehead, 1985; Nelson, 2006; FAO, 2007). This situation may have been exacerbated by the lack of basic biological data and difficulties in identification and taxonomy, and the habit of mixing of different species in the same schools thereby affecting resource harvesting particularly in the warmer tropical and sub-tropical regions (Whitehead, 1985; AIMS, 1999).

Tropical clupeoid fishes, as has been noted, are of prime importance to subsistence and commercial fisheries in many tropical areas such as India, Africa and South East Asia. For

example, the catch of sardine species in the South and East Asian region was about one and a half million tonnes in 1983 and provided the major source of protein for the inhabitants (AIMS, 1999). Within this region, there has been an increasingly substantial landing of clupeoids primarily for fish meal and fish oil for the livestock and rapidly expanding aquaculture industries, and also for food security (FAO, 2007; Alder *et al*, 2008). FAO (2009) reported that fisheries activities in this region (Western Indian Ocean through to the North Western Pacific) has been and are still being fully exploited. The Indian oil sardine (*Sardinella longiceps*), for example, has been the most important clupeoid fishery of the western Indo-Pacific region, from the Gulf of Aden to Bay of Bengal, into the Indonesian archipelago to the Philippines (Longhurst, 1971; Whitehead, 1985). Other species are also fished within this region but provide a smaller fishery or are caught as by catches (Longhurst, 1971; Whitehead, 1985; Raja, 1985; Shifat, 2003; FAO, 2009). It is therefore hardly surprising that some of these species are almost fished into extinction in some areas of this region (Blaber *et al*, 1996; Blaber *et al*, 2003; Wang, 2003).

### **7.5.2 Fly River fishery**

The lack of data on clupeoid biology and fishery is reflected in the Fly River system where very little is known about the biology of most species inclusive of *N. papuensis*. *Nematalosa papuensis* is now being targeted for development as a fish meal source for the feed industry in the Fly River region as is the case for most herring fishery within this region. Any fisheries based on a healthy population of *N. papuensis* must be based on a proper understanding of its biology. This is because the maintenance of its abundance and biomass is also important for the survival of fisheries centered on larger species such as the barramundi (*Lates calcarifer*), the black bass (*Lutjanus goldiei*), and the larger catfish species that prey on it. *Nematalosa papuensis* also plays a significant role as an abundant protein source for the Fly River communities because of the ease in capturing them using small net mesh sizes.

In other tropical lake systems fisheries has had a long history, like that of Lake Tanganyika (between, Burundi, Tanzania, Democratic Republic of Congo, and Zambia) clupeid and latid fishery, which became more formalized in the 1950s-1960s, but plagued with unreliable statistics (Munyandorero, 2002). In comparison the barramundi and black bass fishery in the Fly River system only started in the 1980s (Department of Agriculture, 1991), the Fly River system does not as yet have any clupeid fishery industry. In anticipation of a herring fishery establishment, Wilson (1992) used figures of yield per unit area from Indian systems to estimate a yield of 18 - 33 thousand tonnes per year from the Fly River system (Table 7.1). The Fly system, however, is likely to have a lower sustainable yield than Indian systems, which have many more fish species, with *N. papuensis* comprising about 40% of the catch (Chapter 3). Based on this assumption, Hortle & Storey (2006) estimated the yield of *N. papuensis* at 25% of the total yield estimated from Indian

systems to give an estimated range of 5 - 18 thousand tonnes per year for the Fly system (Table 7.2).

By way of comparison, it is estimated that the Sepik/Ramu floodplains of Northern PNG, has a fishery production of approximately 8,300 tonnes per year, which is based largely on the gill netting of introduced species (Department of Agriculture, 1991). This certainly shows that the potential to develop the Fly River fisheries resource is there as the natural fisheries resource of the Western Province and the Fly River system is far greater than that of the Sepik and Ramu combined

**Table 7.1 Estimates of Maximum Sustainable Yield (MSY) of *N. papuensis* for the Fly River system from Wilson (1992)**

Habitat	Area (km <sup>2</sup> )	Yield (tonnes/km <sup>2</sup> /year)		Total yield (tonnes/year)	
		Low	High	Low	High
Blocked Valley Lakes	245	12	20	2,940	4,900
Oxbow Lakes	122	12	20	1,464	2,440
Floodplains	2,473	3	5	6,183	12,365
Lake Murray	650	12	20	7,800	13,000
<b>Total</b>	<b>3,490</b>			<b>18,387</b>	<b>32,705</b>

**Table 7.2 Revised estimates of *N. papuensis* productivity (Hortle & Storey, 2006)**

Habitat	Area (km <sup>2</sup> )	Yield (tonnes/km <sup>2</sup> /year)		Total yield (tonnes/year)	
		Low	High	Low	High
Blocked Valley Lakes	245	3	5	735	1,225
Oxbow Lakes	122	3	5	366	610
Floodplains	2,473	0.75	1.3	1,855	3,091
Lake Murray	650	3	20	1,950	13,000
<b>Total</b>	<b>3,490</b>			<b>4,906</b>	<b>17,926</b>

These estimates contrast with the actual yield of the *N. erebi* fishery from a lake area of 730 km<sup>2</sup> of 1.37 tonnes/km<sup>2</sup>/year, or a total of 1,000 tonnes/year (Puckridge & Walker, 1990). It is quite possible, therefore, that a small but substantial *N. papuensis* fishery could be developed in the Fly lakes. The estimated yields are approximate and should be refined by monitoring any fishery as it is established. Such a fishery, however, will have its challenges for long-term sustainability and need to be guided by appropriate management plans, because of current trends in global warming that may deleteriously affect the ecology of the lakes, and because of impacts from resource

developments such as the current mine and new oil and gas developments at the headwaters of the Fly.

The effects of such environmental changes on *N. papuensis* populations, which are at the base of the food chain, could be severe. For example, reported declines in biomass and species abundance in the Fly system (e.g. Swales *et al*, 1999) may be due to higher order species being impacted by loss of prey species such as *N. papuensis*. This may mean that any large-scale fishery based on *N. papuensis* may not be viable, with consequent negative impacts on the barramundi and black bass fisheries.

### **7.5.3 Fishery management**

In terms of fishery management, consideration should be given to the fact that females of the species dominate in the larger size classes, especially in the grassed floodplain lakes, which has implications for capture fishery methodology as well as lake conservation management. A closure of fishery is not possible due to this species' year round spawning habit. If, however, a fishery is considered, then selection of fishing gear, example the introduction of maximum mesh size (Pet *et al*, 1996) should aim to prevent exploitation of larger breeding females. Alternatively, a closure fishery may also be introduced to preclude fishing in certain lakes during the year while others are fished, which has been shown have a positive effect on commercial species in freshwater lakes (e.g. Silvano *et al*, 2009). However for this option to work in the Fly River communities, it is recommended that alternative livelihood-sustaining activities or employment is created for the fishing communities (Charles, 2004).

In addition, the numerical abundance of the juveniles of this species in grassed floodplain lakes add emphasis to the need to consider how best to conserve and manage similar un-impacted lakes.

## **7.6 General conclusion**

This study has described the habitat and natural history and dynamics of *N. papuensis* in the lakes of a large tropical river system that is currently being degraded by mine tailings and waste rock dumping in its headwaters. The study has demonstrated that generally habitat quality in the mine-impacted lakes is degraded compared to that of non-mine impacted lakes. The impacts of mining activities, as well as perceived consequences of climate change have, and would result in negative inputs from both the terrestrial aquatic environments into the different habitats, altering the quality and therefore the biology of *N. papuensis* and other species. The impacts as noted elsewhere in this thesis would initially affect flora and fauna at the lower levels of organization and then increase in complexity until fully manifested in whole ecosystem changes. A summary of this synthesis is shown in Figure 7.1. A summary of how the results of this study on *N.papuensis* fit into this holistic impacted ecosystem synthesis is as listed in Table 7.3 below. The findings of this thesis

have shown that sedimentation, metals, and possibly, climate induced impacts on this system have already had negative impacts on the habitats. This has now resulted in negative impacts on the biology of aquatic biota, changes which are manifested in alterations in community and population structure of keystone ecosystem health indicator species such as *N.papuensis*,

**Table 7.3 Summary of effects of ecosystem changes on *Nematalosa papuensis* and other aquatic biota**

<b>Primary Ecosystem stressor</b>	<b>Effect</b>	<b>Initial environment impact</b>	<b>IntermmEDIATE impacts</b>	<b>Ultimate impacts</b>
Sedimentation due to waste rock, tailings/erosion & increased rainfall	Prolonged suspension in water column	Decrease in algal productivity Negative effect on visual feeders	Decrease in juvenile fish; altered floodplain fish community	Negative impact on food security; nutritional deficiencies; alteration of population and community structure & fishery
Chemicals	Metals in water column & bed sediments	Result in dissolved & bioavailable metals in water column, interstitial water & floodplain drainage	Leads to toxicity to aquatic biota; cellular & physiological effects; reduction in algal productivity; impacts on growth & reproduction; toxicity to fish embryos; loss of algal feeding fish & reduction in fish spawning success; reduction in predatorial species	Collapse of artisanal fishery; ecological & behavioural changes & ultimately alteration in population & community structure & fishery
Climate change	Increased temperatures  Increased intensity of <i>El Nino</i> droughts	Reduced dissolved oxygen; increase in dissolved & bioavailable metals  Reduced river & stream flows; reduced organic & inorganic input into lakes	Increased toxicity of chemicals to aquatic biota; increased acidity; reduced algal & planktonic production; fish respiration & feeding affected; negative impacts on planktivores, reproduction, & fish diversity/biomass	Collapse in artisanal & commercial fishery

This alteration in habitat quality has had a significant effect on the growth and reproduction of *N. papuensis* as it is a pelagic spawner with unattached eggs (Growth, 2004) which require good water quality including suitable levels of dissolved oxygen. This study has shown that reproductive fitness of this species is less in mine-impacted and forested lakes compared to unimpacted and grassed floodplain lakes, as shown by negative correlation of GSI against Cu levels.

Even though *N. papuensis* is a highly abundant species, this study indicates that degradation in habitat quality has caused long-term decline in its biomass in the mine-impacted lakes. The diet of *N. papuensis* has been similarly affected by mine impacts since it is mainly a pelagic or benthic phytoplankton/algae feeder. This now limits its foraging range and confines it mainly to the grassland habitats where there is greater productivity under lower mine impacts.

This study has also noted that future compounding impacts of climate change and mining impacts can be very challenging for the recovery of this river system and its fishery, so there is a need to establish a lakes management program outside of the Fly River corridor that can act as refuges for gradual replenishment of the Fly River system after mine closure.





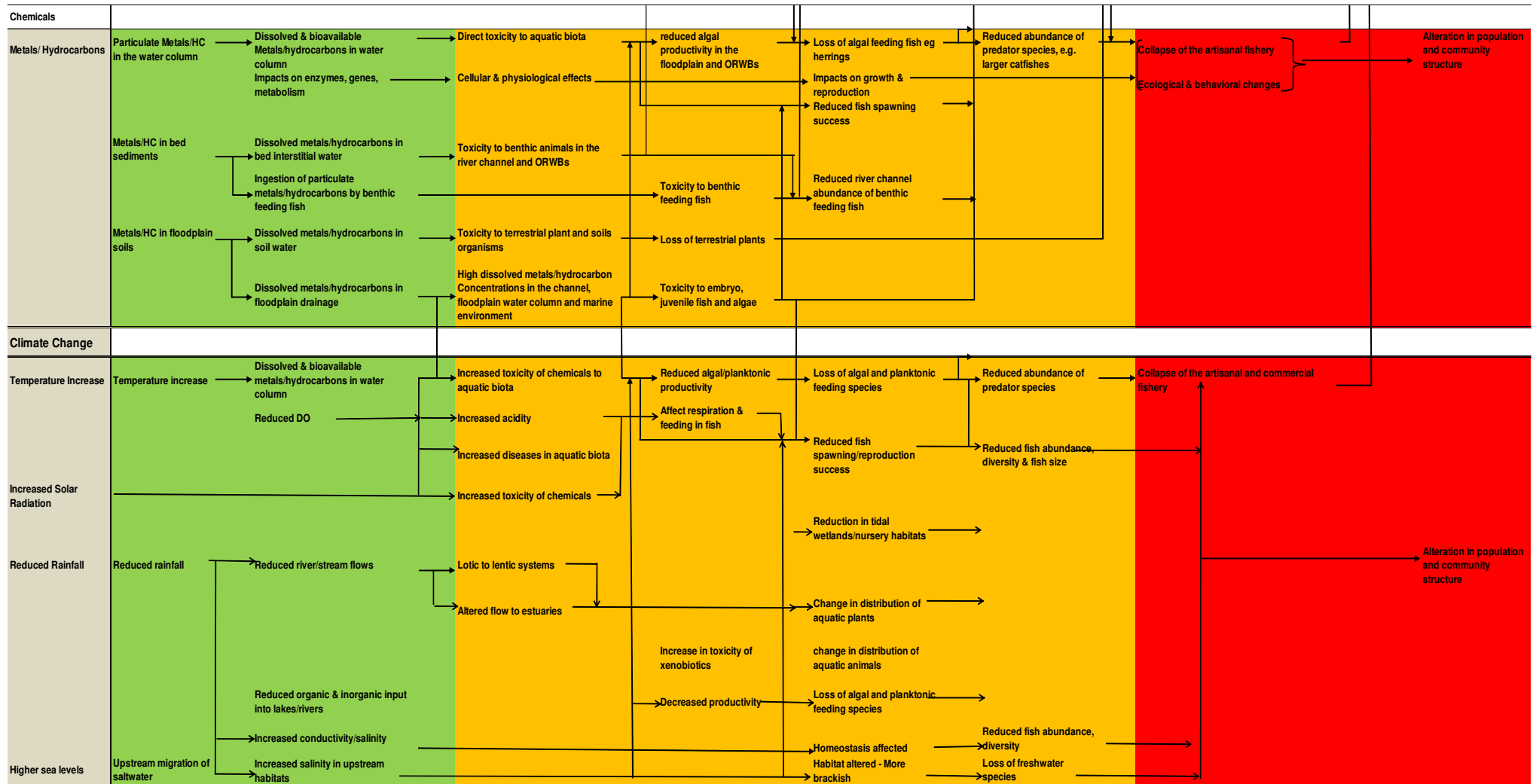


Figure 7.1 Conceptual summary of current impacts on the mine impacted lake habitats of the Fly River system. Colouring is for ease of following (Modified from Timperley, 1999)

Future research on the Fly River system should include work to understand the compounding effects of mining and climate change on fishery ecology; the system-wide effects of xenobiotics on primary consumers; and investigations of the natural history of other key species of fish and macroinvertebrates in this system. This study is only the second species-specific study of the life history of Fly River fishes, the first being that of the barramundi (*L.calcarifer*) undertaken by Government scientists in the early 1980s (Moore, 1980; Moore, 1982; Reynolds & Moore, 1982), despite there being over 120 different fish and macroinvertebrate species in this river system – the largest wetland system in Australasia and the south Pacific. As a result of the government work, barramundi biology is the best understood of any Papua New Guinean freshwater fish species. The summary table (Table 7.3) shows what is now understood about *N. papuensis* in comparison.

**Table 7.4 Summary of biological information on *N.papuensis* and *L.calcarifer* in the Fly and Australia**

Aspect	<i>Nematalosa papuensis</i>	<i>Lates calcarifer</i>
Habitat	Residential in ORWBs <sup>1</sup>	Catadromous, from estuary to river <sup>6</sup>
Spatial and temporal distribution	Juvenile to adult residential populations in ORWBs <sup>2</sup>	Juveniles live in estuary, and river populations are 3-4 year olds <sup>7</sup>
Population structure (	Short lived, fast growing. Maximum age is possibly 3 -4 years <sup>3</sup>	Long lived, at 3-4 years they migrate up river <sup>8</sup>
Diet	Pelagic and detrital planktivore <sup>4</sup>	Ambush predators in river and ORWBs. <i>N. papuensis</i> part of their diet <sup>9</sup>
Reproduction	All year round, no sex change <sup>5</sup>	Seasonal, from October to February and protandrous hermaphrodites <sup>10</sup>

1). Chapter 2, OTML Reports & unpub. data;

2). Chapter 3; OTML Reports & unpub. data

3). Chapter 4

4). Chapter 5

5). Chapter 6

6) Moore, 1980; Moore & Reynolds, 1982; Davis, 1986; Bishop *et al*, 2001

7). Moore, 1980; Davis, 1986; Bishop *et al*, 2001

8). Moore, 1980; Davis & Kirkwood, 1985; Bishop *et al*, 2001

9). Davis, 1985; Kare, 1991; Bunn *et al*, 1999, Bowles *et al*, 2001, Bishop *et al*,

2001; WRM, 2006; Storey & Yarrao, 2009;

10). Moore, 1980; Moore, 1982; ; Davis, 1985;; Bishop *et al*, 2001

Despite this knowledge of the fish biology of *N. papuensis*, of greatest concern now is that, the future of this massive wetland system is under serious threat from both mining and climate change impacts, as stated elsewhere. The impacts of the environment reflected in the biology of this keystone species is indicative of the extent to which other species will have already been affected. There is now a serious need to research further the future implications of these alterations and predictions on fisheries ecology, proposed fishery developments, and community livelihood along this river system.

It is therefore highly recommended that the current waste management, or rather the lack of it,

resulting in the practice of direct dumping of mining and milling wastes into river systems should be replaced by better well researched alternatives with a view to adapting different methods of mine and mill waste management.

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## Appendix 2.1 Ok Tedi Mining Limited Fologian rainfall data 1984-2008; and rainfall at other 2 sites (Taranaki & Southern dumps)

### a) Fologian – Data used in this study

FOLOMIAN (Ok Ningi dam area)

1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008
505.8	820.7	821.8	729	723.3	907.2	1061.1	626.2	893.2	754	797	549.8	888.1	772	574.1	571.2	509.2	919.0	841.4	678.6	725.3	831.9	766.5	722.0
595.6	666.8	683.0	631.7	711.6	774.4	714.2	747.5	704.6	574.0	600.4	607.0	408.2	922.0	626.4	365.6	607.0	545.4	803.6	776.6	741.6	743.4	544.2	653.7
776.2	735	796.5	636.5	605.8	446.6	666.2	711.8	589.4	632.2	1091.8	607	329.9	839.4	842.8	579.8	608.4	813.2	1010.0	691.5	693.7	655.1	711.2	712.9
514.2	877.5	653.5	576.3	575.9	521.4	783.8	746.2	944.3	747.4	744.2	659.2	740.8	779	587	645.6	942.4	641.0	726.8	698.8	745.7	674.9	607.9	574.5
624.0	615.2	589.4	600.4	712.6	528.3	728.0	605.8	703.6	866.0	613.2	502.4	393.4	662.0	541.2	554.4	540.2	605.2	607.2	653.3	979.0	543.6	659.9	756.3
559.8	848.6	741.2	796.6	414.0	713.4	460.4	703.6	1190.0	791.5	926.0	770.2	270.2	474.8	622.6	809.8	913.4	710.0	137.8	784.5	577.2	898.6	596.1	937.0
778.2	648.8	493.6	761.2	543.4	606.6	824.2	848.3	1024	877	675	605.3	655.6	564	541.2	778.8	823.0	670.0	1126.2	807.3	881.7	617.7	708.1	565.0
787.3	466.7	625.8	576.6	959.2	507.0	933.8	487.0	388.6	1436.0	802.6	560.6	96.2	609.0	1017.6	1059.0	806.8	616.0	672.8	608.8	881.0	750.0	917.4	746.9
747.2	684.2	592.3	637.0	614.6	783.7	385.4	374.2	431.7	570.8	721.2	623.6	96.2	647.0	656.4	675.0	716.4	317.2	966.6	848.1	791.8	964.6	922.4	736.9
627.2	450.6	663.5	685.0	547.2	840.8	758.6	551.2	287.2	574.0	1005.8	985.0	159.2	618.4	541.6	753.0	738.4	602.2	659.4	299.5	716.4	472.6	776.7	582.2
581.4	491.4	547.5	462.0	812.6	452.2	598.4	354.4	584.3	552.2	452.9	505.6	357.2	677.8	593.0	753.6	697.0	610.8	672.8	576.2	706.3	517.4	507.5	531.8
797.6	360.2	1024.8	472.2	643.4	565.2	741.6	848.8	732.9	443.6	796	536.2	590.2	962.7	690.2	516.8	551.4	464.8	0.0	566.1	694.1	496.5	805.8	752.8
<b>7894.50</b>	<b>7665.70</b>	<b>8232.90</b>	<b>7564.50</b>	<b>7863.60</b>	<b>7646.80</b>	<b>8655.70</b>	<b>7605.00</b>	<b>8473.80</b>	<b>8818.70</b>	<b>9226.10</b>	<b>7511.90</b>	<b>4985.20</b>	<b>8528.10</b>	<b>7834.10</b>	<b>8062.60</b>	<b>8453.60</b>	<b>7514.83</b>	<b>8224.60</b>	<b>7989.30</b>	<b>9133.78</b>	<b>8166.29</b>	<b>8523.67</b>	<b>8272.0</b>

### b) Southern Dumps

1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008
1153.5	493	890	767.9	748.8	636.8	632.6	739.6	646.4	747.4	719.0	988.4	851.8	747.3	844.0	936.0	787.0	689.0
754.2	778	683.2	701.5	601.3	624.8	490.8	1062.6	697.4	409.6	686.6	642.2	895.8	841.7	821.0	856.6	620.0	668.0
837.4	757.7	608.4	690.4	842.9	669.3	408.5	854.5	876.8	601.8	660.6	872.2	1116.6	858.7	720.1	736.2	840.0	810.0
927.4	821.4	1044.2	756.8	858.1	643	700.1	774.8	617.6	645.6	1003.8	786.2	705.2	680.8	859.8	735.0	802.0	818.0
924.6	738	882.6	937.6	781.8	465	318.2	653	472.4	537.2	620.8	699.6	841	732.4	1123.1	615.0	706.0	964.0
666	1015.8	1385.7	968.8	1386.8	779.2	344.2	473.2	628	829.8	1155.6	934.4	261.8	1036.1	778.6	1067.0	1201.0	1364.0
1071.4	1215.2	1384.9	1171.4	978.7	508	746.4	590.4	990.2	685.6	1040.4	844.4	1390.8	968.5	1147.4	797.0	1301.0	888.0
1121.2	595.2	460.3	1834.5	942.2	547.6	112.2	845.2	1133.2	1147.2	1037.6	882.0	863.8	639.3	1129.7	1027.0	1301.0	1172.0
404	485.2	640.1	671.8	873.4	497.2	117.2	765.6	767.2	779.2	918.8	461.4	1038	1089.2	962.4	1299.0	1207.0	928.0
713	630	449.7	518.6	1002.4	907.3	92.8	711.4	613	735.8	819.8	600.8	872.2	384.4	765.2	680.0	1011.0	729.0
606.8	392	664.9	567.7	559.2	388.8	255.6	800.4	680	689.4	972.4	745.2	776.5	771.8	764.4	523.0	583.0	652.0
720	882.6	802.2	437.2	798.2	615.2	552.6	1050	709.4	677.8	705.5	505.2	737.9	732.1	817.3	0.0	838.0	826.0
<b>9899.5</b>	<b>8804.1</b>	<b>9896.2</b>	<b>10024.2</b>	<b>10373.8</b>	<b>7282.2</b>	<b>4771.2</b>	<b>9320</b>	<b>8831.6</b>	<b>8486.4</b>	<b>10340.9</b>	<b>8962</b>	<b>10351.4</b>	<b>9482.3</b>	<b>10733.0</b>	<b>9271.8</b>	<b>11197.0</b>	<b>10508.0</b>



c) Taranaki

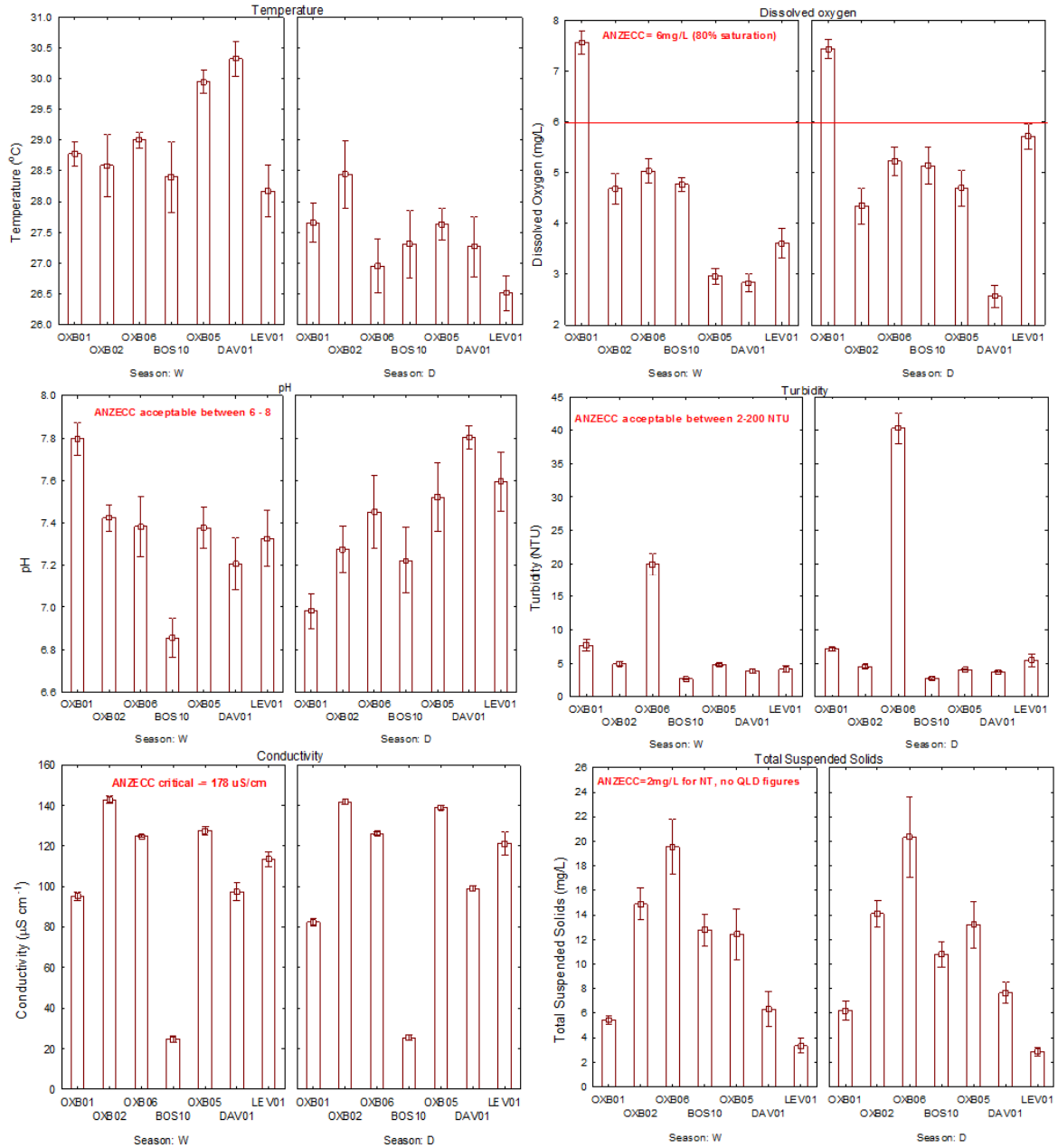
TARANAKI RAINFALL (mm)

1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008
1086.4	582.2	1012.0	727.4	717.0	663.5	825.0	708.0	702.0	747.4	674.8	904.2	806.6	650.2	778.6	792.0	848	0.0
842.0	781.4	797.0	640.9	575.2	731.2	458.2	865.6	733.0	401.8	680.2	558.6	803.6	763.4	755.3	766.4	654	0.0
763.8	827.4	564.4	621.1	833.1	680.1	504.6	834.2	997.2	629.4	652.6	747.2	998.8	747.4	706.3	683.0	810	0.0
843.2	785.3	957.4	737.2	782.1	866.2	763.6	809.0	677.0	627.6	987.8	731.2	631.4	747.2	840.7	689.0	762	0.0
799.0	728.2	761.4	902.1	693.4	498.0	400.1	699.2	606.6	593.4	631.4	606.0	624.2	587.2	1035.2	569.0	604	0.0
609.4	896.8	1433.9	887.9	1128.0	932.8	355.0	548.8	660.2	870.6	1029.8	804.8	173.4	759.4	687.3	944.0	657	0.0
985.8	1141.6	1217.1	1000.5	885.0	687.0	799.0	639.4	1039.0	794.4	959.8	658.5	1094.0	832.3	1000.8	745.0	957	0.0
1099.8	622.0	461.0	1728.7	914.8	693.6	125.2	842.6	1290.4	1129.8	937.0	632.7	690.0	551.35	991.6	957.0	1137	0.0
415.4	426.0	509.9	598.7	832.0	714.2	84.2	777.0	814.0	648.2	711.4	377.4	927.4	842.4	865.8	1149.0	1058	0.0
742.6	634.4	444.0	585.3	1116.0	1078.6	105.8	646.8	628.0	703.6	711.6	541.8	691.0	302.9	690.2	528.0	836	0.0
534.0	367.2	545.1	502.6	550.8	485.6	305.6	754.4	802.4	680.8	739.5	566.0	585.0	693	695	445.0	570	0.0
734.6	1063.4	748.0	489.2	911.8	573.6	567.2	1004.2	630.6	678.4	614.4	481.5	624.0	651.75	761	0.0	731	0.0
<b>9456.00</b>	<b>8855.90</b>	<b>9451.20</b>	<b>9421.60</b>	<b>9939.20</b>	<b>8604.40</b>	<b>5293.50</b>	<b>9129.20</b>	<b>9580.40</b>	<b>8505.40</b>	<b>9330.30</b>	<b>7609.85</b>	<b>8649.40</b>	<b>8128.50</b>	<b>9807.80</b>	<b>8267.4</b>	<b>9624.0</b>	<b>0.0</b>

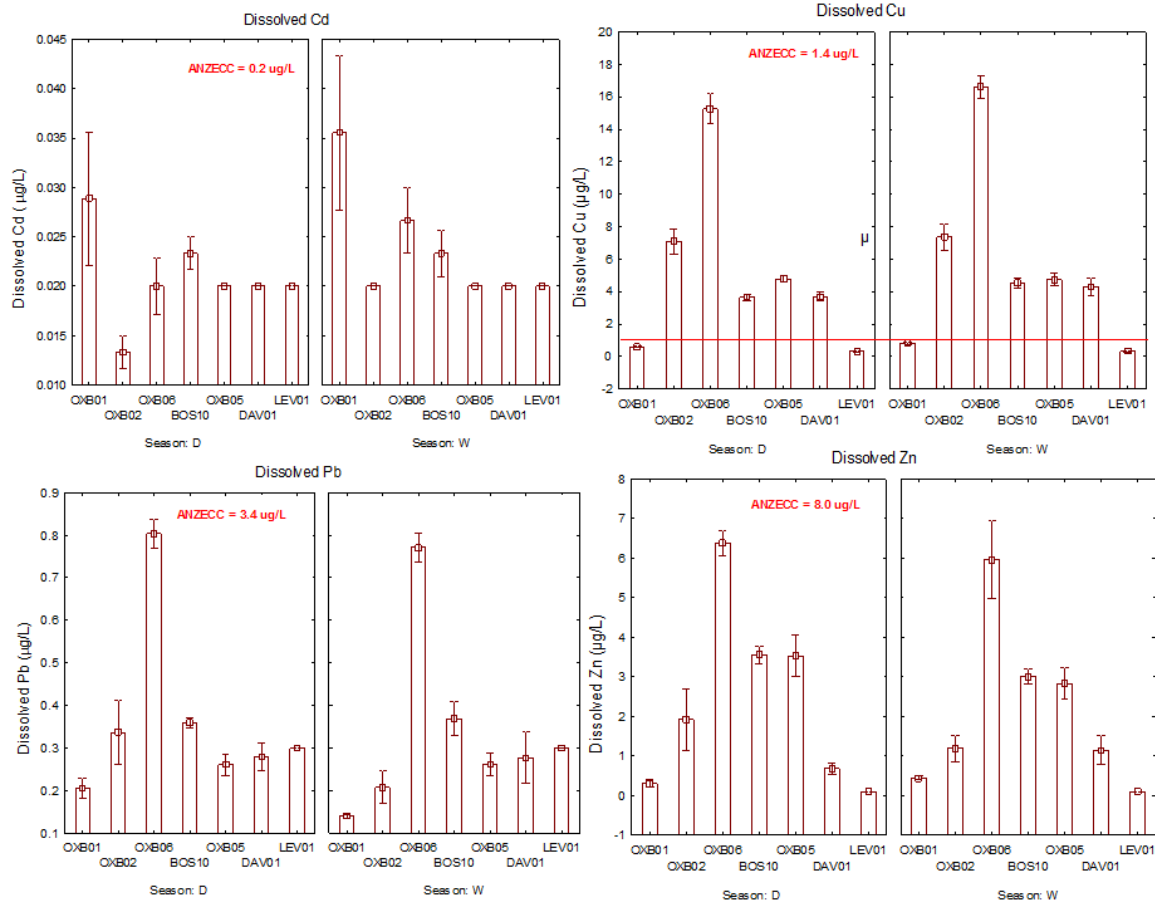
## Appendix 2.2 ORWB Macrophyte list

Family	Species	Site recorded from
Azollaceae	<i>Azolla pinnata</i> R. Br.	OXB02, OXB06, BOS10, OXB05, DAV01
Araceae	<i>Pistia stratiotes</i> L.	OXB05, DAV01
Ceratophyllaceae	<i>Ceratophyllum demersum</i> L.	BOS10, OXB05, DAV01
Nymphaeaceae	<i>Nelumbo nucifera</i> Gaertn	OXB06, BOS10, OXB05, DAV01, LEV01
	<i>Nymphaea nouchali</i> Burm.f.	BOS10, DAV01, OXB05, LEV01
	<i>Nymphaea pubescens</i>	OXB05, DAV01, BOS10
	<i>Nymphaea violacea</i>	DAV01, OXB05, BOS10
	<i>Nymphaea macrosperma</i>	DAV01, BOS10
Lemnaceae	<i>Lemna perpusilla</i> (duckweed)	DAV01
Parkeriaceae	<i>Cerapteris thalictroides</i> (L.) Brongn.	OXB05
Convolvulaceae	<i>Ipomea aquatica</i> Forsk.	OXB02, OXB06, BOS10, OXB05, DAV01, LEV01
Cyperaceae	<i>Actinoscirpus grossus</i> (L. f.) Goetgh. & Simpson	DAV01, OXB05
	<i>Cyperus bifax</i> C.B. Clarke	DAV01, OXB06
	<i>C. brevifolius</i> (Rottb.) Hassk.	BOS10, DAV01, OXB05
	<i>C. diffusus</i> Vahl.	BOS10, DAV01, OXB05, LEV01
	<i>C. odoratus</i> L.	DAV01
	<i>Fimbristylis dichotoma</i> (L.) Vahl	BOS10
	<i>F. globosa</i>	BOS10
	<i>Fuirena ciliaris</i> (L.) Roxb.	OXB06, BOS10, OXB05, DAV01
	<i>Schoenus</i> sp.	BOS10, OXB05, DAV01
	<i>Scleria ciliaris</i> Nees.	BOS10, DAV01
Hydrocharitaceae	<i>Hydrocharis dubia</i> (Bl.) Backer	BOS10, OXB5, DAV01
	<i>Hydrilla verticillata</i> (L.f) Royle	OXB06, BOS10, DAV01, OXB05
	<i>Vallisneria natans</i> (Lour.) Hara	DAV01
Najadaceae	<i>Najas indica</i> (Willd.) Cham.	OXB06
Poaceae	<i>Ischaemum polystachyum</i> L.	OXB06, BOS10, DAV01, OXB05, OXB06, BOS10, DAV01, OXB05, LEV01
	<i>Leersia hexandra</i> Sw.	LEV01
	<i>Echinochloa praestens</i>	BOS10, OXB5, DAV01, LEV01
	<i>Echinochloa stagnina</i> (Retz.) P.Beauv.	OXB06, BOS10, DAV01, OXB05
	<i>Hymenachne amplexicaulis</i> (Rudge) Nees	OXB06, BOS10, DAV01, OXB05
	<i>Hymenachne acutigluma</i>	OXB06, BOS10, DAV01, OXB05
	<i>Ophiuros tongcalingii</i> (Elm.) Henr.	BOS10
	<i>Isachne globosa</i> (Thunb.) Kuntze	BOS10, DAV01
	<i>Phragmites karka</i> (Retz) Trin.	OXB02, OXB06, BOS10, OXB05, DAV01, LEV01
	<i>Panicum auritum</i>	DAV01
	<i>Setaria pallide-fusca</i>	BOS10
	<i>Oryza rufipogon</i>	BOS10, DAV01, OXB05
Flagellariaceae	<i>Hanguana malayana</i> (Jack) Merr.	BOS10, OXB05
Lentibulariaceae	<i>Utricularia aurea</i>	OXB05
Menyanthaceae	<i>Nymphoides aurantiaca</i>	BOS10, OXB05, DAV01
	<i>Nymphoides exiliflora</i>	BOS10, OXB05, DAV01
	<i>Nymphoides geminata</i>	BOS10, OXB05, DAV01, LEV01
	<i>Nymphoides indica</i> (L.) Kuntze	BOS10, OXB05, DAV01, LEV02
Onagraceae	<i>Ludwigia hyssopifolia</i> (G.Don) Exell	OXB06, BOS10, OXB05, DAV01
	<i>Ludwigia adscendens</i> (L.) Hara	BOS10, DAV01
Polygonaceae	<i>Polygonum attenuatum</i> R.Brown	OXB06, BOS10, OXB05, DAV01
	<i>Polygonum barbatum</i>	DAV01
Pandanaceae	<i>Pandanus</i> Sp.	OXB01, OXB02, OXB06
Potamogetonaceae	<i>Potamogeton</i> sp.	BOS10, DAV01
	<i>Potamogeton pussilus?</i>	BOS10, DAV01
Psilotaceae	<i>Psilotum</i> sp.	DAV01
Scrophulariaceae	<i>Limnophila aromatica</i> (Lamk) Merr.	DAV01
	<i>Limnophila indica</i> (L.) Druce	BOS10, DAV01

**Appendix 2.3 Water Quality Variables over each season and where appropriate indication of the critical or acceptable levels as per Australian and New Zealand Water Quality Guidelines**



**Appendix 2.4 Average dissolved metals in water column at all sites from Fly upstream to most downstream site, with highest critical acceptable levels indicated**



**Appendix 3.1 Summary of catches of *N. papuensis* during this study.**

Habitat type	Location	Site	Wet season (3 trips)		Dry season (3 trips)		Mean site totals		Mean location totals	
			Mean		Mean		Numbers	Biomass (kg)	Number caught	Biomass (kg)
			Mean Number collected	Mean biomass caught(kg)	Mean Number collected	Mean biomass caught (kg)				
		1	8.00	0.62	17.00	0.11	25.00	0.73		
		2	7.00	0.42	13.00	0.81	20.00	1.23		
	OXB01	3	6.30	0.36	9.00	0.53	15.30	0.89	60.30	2.85
		1	41.67	4.07	87.33	7.58	129.00	11.65		
		2	55.67	3.67	114.67	7.56	170.34	11.24		
	OXB02	3	32.00	2.42	143.33	9.66	175.33	12.07	474.67	34.96
		1	131.33	4.93	105.33	5.02	236.66	9.95		
		2	119.67	7.38	124.67	5.86	244.34	13.24		
<b>Forested oxbow</b>	OXB06	3	270.33	6.00	186.33	6.17	456.66	12.17	937.66	35.36
		1	131.67	7.57	338.33	18.71	470.00	26.28		
		2	97.33	6.49	395.33	22.50	492.66	28.99		
<b>Grassed oxbow</b>	OXB05	3	106.00	3.97	369.33	19.44	475.33	23.41	1437.99	78.68
<b>Blocked Valley</b>		1	390.33	20.54	301.00	15.09	691.33	35.63		
<i>Mixed (forested/grassed)</i>		2	384.67	19.07	345.67	19.95	730.34	39.02		
	BOS10	3	257.33	12.56	194.00	8.87	451.33	21.43	1873.00	96.07
		1	86.00	6.92	140.00	9.25	226.00	16.17		
<i>Grassed</i>		2	75.67	4.66	120.00	8.87	195.67	13.53		
	DAV01	3	44.67	2.86	104.00	7.06	148.67	9.92	570.34	39.62
<b>Mixed (forested/grassed) oxbow</b>		1	157.67	6.87	173.00	7.66	330.67	14.53		
		2	146.33	7.04	224.67	10.79	371.00	17.83		
	LEV01	3	251.33	8.28	223.33	7.05	474.66	15.34	1176.33	47.69
	<b>Totals</b>		2800.97	136.70	3729.32	198.53	6530.29	335.23	6530.29	335.23

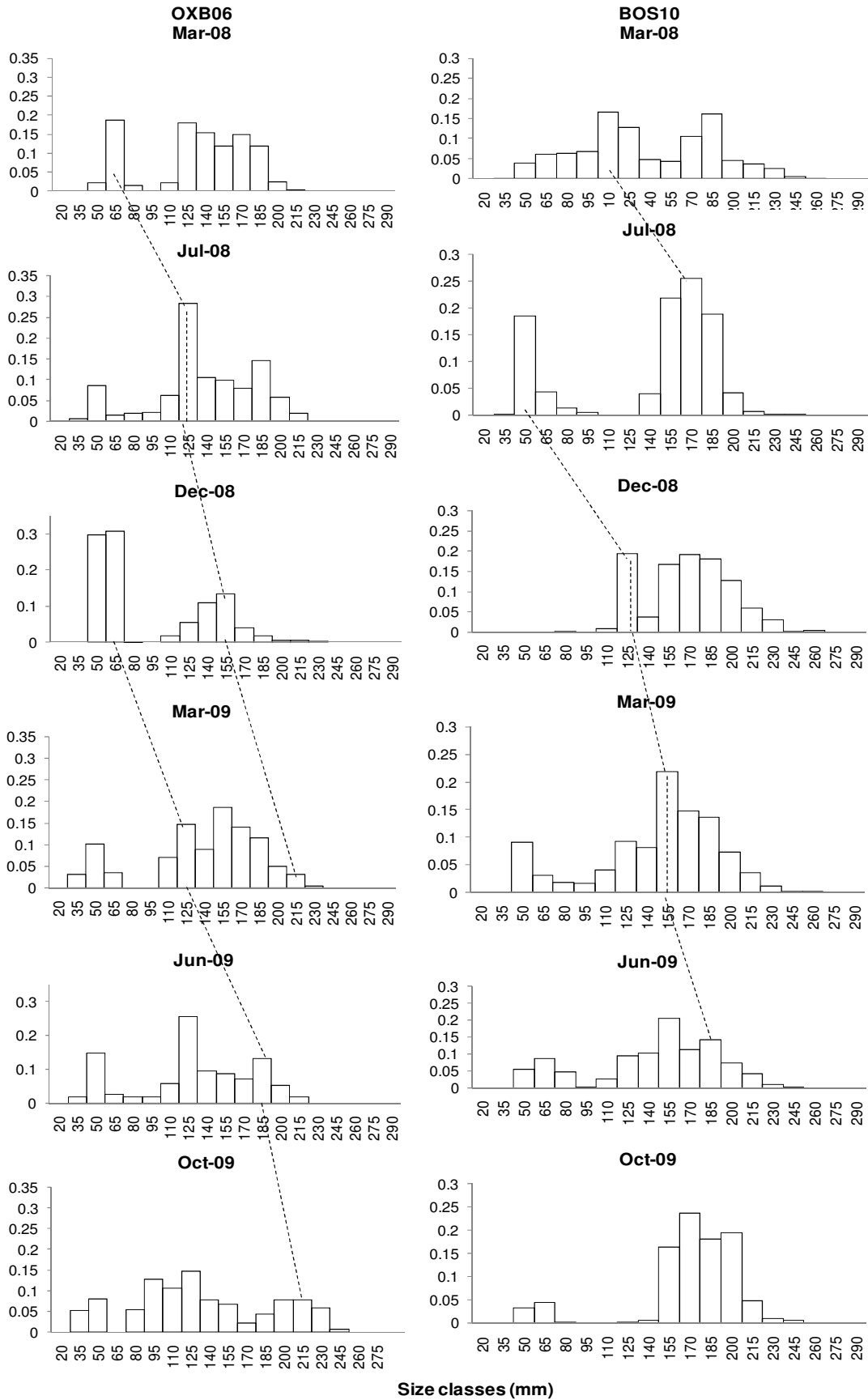
**Appendix 3.2 Summary of historical *N. papuensis* catches. Shaded boxes indicate that only 1 sampling trip was done during that season. The 2008/2009 data are italicised.**

Habitat type	Location	Year	Number of sampling trips	Wet season		Dry season	
				Number collected	Biomass caught (kg)	Number collected	Biomass caught (kg)
		1989	1			29	2.327
		1990	3	180	7.228	587	42.74
		<i>2008</i>	3	39	<i>2.716</i>	<i>20</i>	<i>1.36</i>
	OXB01	<i>2009</i>	3	<i>25</i>	<i>1.495</i>	<i>88</i>	<i>5.365</i>
		1990	3	117	5.752	492	43.762
		<i>2008</i>	3	<i>185</i>	<i>16.502</i>	<i>225</i>	<i>16.285</i>
	OXB02	<i>2009</i>	3	<i>90</i>	<i>6.819</i>	<i>436</i>	<i>26.432</i>
		1993	3			3015	155.352
		1994	3	1450	65.535	1651	65.785
		1995	4	894	57.464	2111	111.884
		1996	3	845	49.612	359	28.089
		1997	2	202	14.269	151	9.266
		1998	4	497	28.575	696	45.601
		1999	3	436	27.897	213	14.845
		2000	3	363	28.077	292	36.287
		2001	2	385	35.528		
		2002	1	189	19.19		
		<i>2008</i>	3	<i>771</i>	<i>41.282</i>	<i>406</i>	<i>18.796</i>
<b>Forested oxbows</b>	OXB06	<i>2009</i>	3	<i>188</i>	<i>11.704</i>	<i>638</i>	<i>32.085</i>
		1989	2			243	27.942
		1990	3	114	13.006	1225	123.274
		1991	2			1426	162.157
		1992	4	227	30.758	4683	378.924
		1993	4	1372	106.746	1405	131.794
		1994	3	7	0.01	12	0.954
		1995	4	618	46.115	987	47.833
		1996	4	639	40.688	639	44.022
		1997	2	288	20.943	229	19.193
		1998	4	410	46.629	496	49.156
		1999	3	304	30.477	190	24.568
		2000	4	292	40	335	52.688
		2001	2	214	35.101		
		2002	1	190	28.182		
		<i>2008</i>	3	<i>440</i>	<i>38.109</i>	<i>1595</i>	<i>94.857</i>
<b>Grassed oxbow</b>	OXB05	<i>2009</i>	3	<i>342</i>	<i>15.023</i>	<i>1574</i>	<i>85.74</i>
<b>Blocked Valley</b>		1988	3	657	16.982	1134	95.976
		1989	3	590	62.133	539	67.28
		1990	4	478	64.976	813	110.895
		1991	2	139	0.0322	550	82.087
		1992	3	801	138.469	773	67.466
		1993	3	553	18.662	2266	202.918
		1995	3	246	11.409	99	7.998
		1996	2	237	14.282	139	8.559
		1997	1	99	5.667		
		1998	2			254	19.731
		1999	2	92	7.553	149	15.569
		2000	4	388	35.564	198	24.639
		2001	2	351	43.498		
		2002	1	219	34.483		
		<i>2008</i>	3	<i>1805</i>	<i>102.835</i>	<i>521</i>	<i>32.908</i>
<b>Mixed (forested/grassed)</b>	BOS10	<i>2009</i>	3	<i>728</i>	<i>45.74</i>	<i>1527</i>	<i>97.471</i>
		1988	4	105	6.485	13	805
		1989	3	96	7.365	105	8.112
		1990	3	220	20.402	606	59.562
		1991	2			424	29.764
		1992	3	164	11.535	42	2.233
		1993	3	1086	49.702	63	3.752
		1994	2	104	0.788	1	0.041
		1995	4	28	1.293	632	38.567
		1996	4	348	18.992	359	26.901
		1997	1			71	4.09
		1998	1			4	0.25
		1999	3	90	5.775	57	5.803
		2000	4	171	20.905	267	37.483
		2001	1	95	11.445		
		2002	1	128	19.615		
		<i>2008</i>	3	<i>325</i>	<i>25.277</i>	<i>519</i>	<i>40.215</i>
<b>Grassed</b>	DAV01	<i>2009</i>	3	<i>208</i>	<i>17.907</i>	<i>501</i>	<i>34.104</i>
<b>Mixed (forested/grassed) oxbow</b>	LEV01	<i>2008</i>	3	<i>1053</i>	<i>47.307</i>	<i>233</i>	<i>13.299</i>
		<i>2009</i>	3	<i>406</i>	<i>18.792</i>	<i>1313</i>	<i>61.947</i>

#### Appendix 4.1 Size class distribution of *N. papuensis* in all study lakes

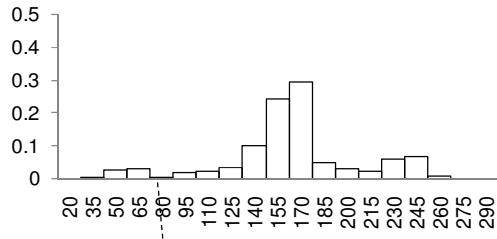
Size class (mm)	OXB01		OXB02		OXB06		BOS10		OXB05		DAV01		LEV01		Percent of population
	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	
20															
35	0	0	0	0	2.09	0.48	0.04	0.07	0.09	0	0	0.17	0.75	0.68	0.33
50	0	0	0.34	0	11.24	12.50	8.60	4.67	0.51	0.47	1.13	3.31	3.97	6.81	4.68
65	0	0	0.41	0	1.69	20.33	6.78	4.17	1.54	1.40	4.60	4.18	5.26	1.11	4.45
80	0	0	0.07	0	2.73	0.89	3.61	3.54	10.61	1.17	2.69	2.26	9.50	0.56	4.25
95	0	0	0.20	0.57	4.41	0.07	0.79	4.67	6.10	3.50	4.43	0.17	5.48	0.99	3.17
110	0	0.76	2.93	2.84	6.98	2.88	1.47	10.26	3.17	2.33	2.00	1.39	8.86	10.96	5.22
125	0	1.52	3.54	0.85	24.08	13.32	5.23	13.05	9.88	9.22	4.17	9.06	9.93	21.61	10.75
140	2.38	6.06	5.31	6.25	9.39	12.91	5.95	6.16	10.06	6.42	4.08	7.84	17.34	15.60	9.18
155	40.48	35.61	22.80	12.50	8.67	13.46	19.90	11.62	18.98	15.99	21.27	17.42	16.37	20.19	17.14
170	42.86	18.94	20.90	19.32	6.42	11.26	17.41	13.15	12.99	19.72	19.88	18.82	11.43	9.78	14.43
185	11.11	27.27	19.20	21.31	11.72	8.45	17.17	15.00	12.99	23.69	17.97	9.76	6.12	8.54	13.76
200	3.17	5.30	12.53	21.02	6.02	2.27	8.72	6.69	6.35	12.37	7.12	5.92	2.63	2.48	6.70
215	0	3.03	8.92	11.65	3.13	0.89	3.29	4.07	4.20	3.15	5.12	6.45	1.40	0.68	3.72
230	0	0.76	2.18	3.13	1.28	0.21	0.75	2.28	1.90	0.47	2.86	7.49	0.86	0	1.57
245	0	0	0.48	0.57	0.16	0.07	0.28	0.43	0.51	0.12	2.00	4.70	0.11	0	0.52
260	0	0	0.20	0	0	0	0	0.17	0.09	0	0.52	0.70	0	0	0.11
275	0	0.76	0	0	0	0	0	0	0	0	0.17	0.35	0	0	0.03
	100	100	100	100	100	100	100	100	100	100	100	100	100	100	100

**Appendix 4.2 Representation of year-round size cohorts in *N. papuensis***

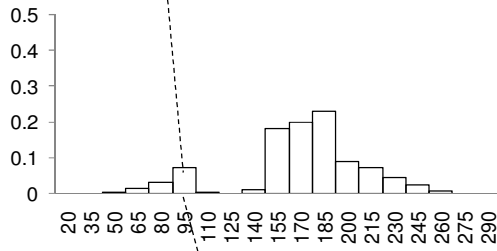




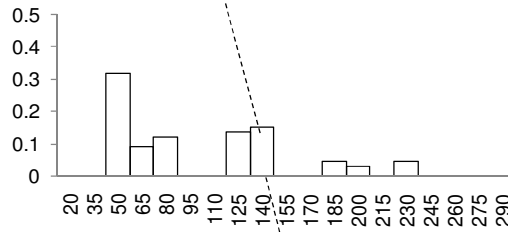
**DAV01  
Apr-08**



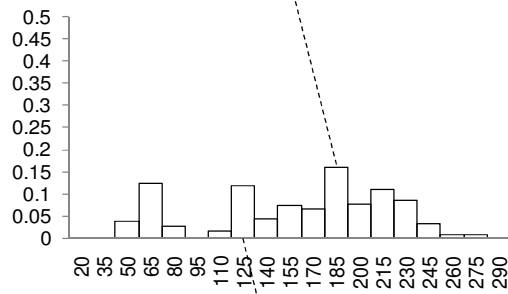
**Jul-08**



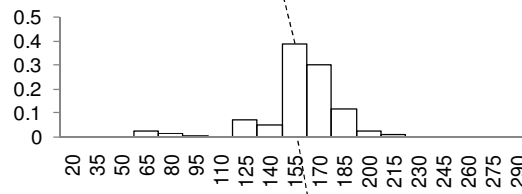
**Dec-08**



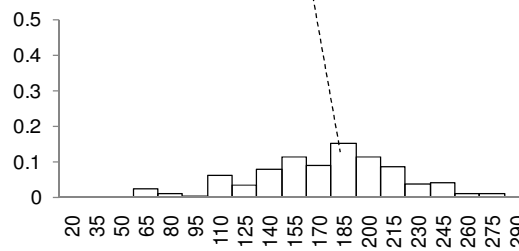
**Mar-09**



**Jun-09**



**Oct-09**



**Size classes (mm)**

**Appendix 4.3. Growth rate of *N. papuensis* cohorts in OXB06, BOS10, and DAV01 during the study period**

<b>Lake</b>	<b>Date 1</b>	<b>Date 2</b>	<b>Size class 1</b>	<b>Size class 2</b>	<b>Number of months</b>	<b>Change in length</b>	<b>Growth rate (mm/month)</b>
OXB06	28-Mar-08	20-Jul-08	65	125	3.75	60	16
	20-Jul-08	07-Dec-08	125	155	4.25	30	7.06
	07-Dec-08	19-Mar-09	155	215	3.25	60	18.46
	01-Dec-08	01-Mar-09	65	125	2.5	60	24
	19-Mar-09	11-Jun-09	125	185	2.5	60	24
	11-Jun-09	25-Oct-09	185	215	4.5	30	6.67
	<b>Mean</b>						16.03
BOS10	28-Mar-08	20-Jul-08	110	170	3.75	60	16.00
	20-Jul-08	07-Dec-08	50	125	4.25	75	17.65
	07-Dec-08	19-Mar-09	125	155	3.25	30	9.23
	19-Mar-09	11-Jun-09	155	185	2.5	30	12
	<b>Mean</b>						13.72
DAV01	5-Apr-08	01-Aug-08	65	95	3.75	30	8.00
	01-Jul-08	02-Dec-08	95	140	5	45	9.00
	02-Dec-08	25-Mar-09	140	185	3.75	45	12.00
	02-Dec-08	25-Mar-09	50	120	3.75	70	18.67
	16-Jun-09	28-Oct-09	155	185	4.5	30	6.67
	<b>Mean</b>						10.87
<b>All lakes mean growth rate per month</b>							<b>13.69</b>

#### Appendix 4.4 Growth rate of *N. papuensis* in ponds

Number of Fish sampled	Month 1	Month 2	Mean Lt 1 (mm)	Mean Lt 2 (mm)	Number of months	Change in length	Growth rate (mm/month)
4	Jul-07	Nov-07	70	172.25	4	102.25	25.56
4	Nov-07	Mar-08	172.25	206.75	4	34.5	8.63
4	Mar-08	Aug-08	206.75	244.5	5	37.75	7.55
4	Aug-08	Mar-09	244.5	270.5	7	26	5.7
<b>Mean growth rate in ponds per month</b>							<b>11.86</b>

**Appendix 5.1 Algal taxa in diet**

Taxa	OXB01	OXB02	OXB06	BOS10	OXB05	DAV01	LEV01
<b>Bacillariophyta (Diatoms)</b>							
<i>Acanthoceras sp</i>				1	1	1	
<i>Cymbella sp</i>				1			1
<i>Cyclotella sp</i>				1			
<i>Flagilaria (Synedra?) sp</i>	1	1	1	1	1	1	1
<i>Gomphonema sp</i>					1	1	1
<i>Tabellaria sp</i>				1	1	1	1
<i>Urosolenia sp</i>						1	1
<i>Aulacoseira sp</i>	1		1	1	1	1	1
<b>Chrysophyta (Golden Brown algae)</b>							
<i>Mallomonas sp</i>			1	1	1	1	1
<b>Chlorophyta (Green alga &amp; Desmids)</b>							
<i>Ankistrodesmus sp</i>		1		1		1	
<i>Closterium sp</i>	1	1	1	1	1	1	1
<i>Cosmarium sp</i>				1	1	1	
<i>Micrasterias sp</i>				1	1	1	
<i>Oocystis sp</i>				1	1	1	
<i>Scenedesmus sp</i>			1			1	
<i>Staurastrum sp</i>				1	1	1	
<i>Ulothrix sp</i>		1			1		1
<i>Stichococcus sp</i>		1			1	1	
<i>Chlamydomonas sp</i>		1					
<i>Pediastrum sp</i>				1		1	
<i>Oedogonium sp</i>		1		1		1	
<i>Bulbochaete sp</i>						1	
<i>Botryococcus sp</i>				1		1	
<i>Spirogyra sp</i>			1	1		1	1
<i>Volvox sp</i>			1	1	1	1	
<b>Cryptophyta (Cryptomonads)</b>							
<i>Cryptomonas sp</i>							
<b>Cyanophyta (Cyanobacteria)</b>							
<i>Anabaena sp</i>		1	1				
<i>Cylindrospermopsis sp</i>	1	1					
<i>Merismopedia sp (blue-green algae)</i>				1		1	
<b>Dinophyta (Dinoflagellates)</b>							
<i>Peridinium sp</i>		1	1	1	1	1	1
<b>Euglenophyta</b>							
<i>Euglena sp</i>	1		1		1	1	1
<i>Phacus sp</i>							1
<i>Trachelomonas sp</i>				1			
<b>Totals</b>	<b>5</b>	<b>10</b>	<b>10</b>	<b>21</b>	<b>16</b>	<b>24</b>	<b>13</b>

Presence/absence data of algae genera found in *Nematalosa* guts from all sites (0 = absence, 1 = presence).

**Appendix 5.2 Guts with and without food during wet and dry seasons**

<b>Lake</b>	<b>Season</b>	<b>With food</b>	<b>Without food</b>	<b>Total</b>
OXB01	D	9	11	20
OXB01	W	15	5	20
OXB02	D	10	10	20
OXB02	W	17	3	20
OXB06	D	6	14	20
OXB06	W	18	2	20
BOS10	D	12	8	20
BOS10	W	20	0	20
OXB05	D	14	6	20
OXB05	W	18	2	20
DAV01	D	15	5	20
DAV01	W	18	2	20
LEV01	D	12	8	20
LEV01	W	19	1	20
<b>Sum</b>	<b>Wet season</b>	<b>125</b>	<b>15</b>	<b>140</b>
	<b>Dry season</b>	<b>77</b>	<b>63</b>	<b>140</b>

**Appendix 5.3 Stable isotope data**

Habitat	Lake	Sex	Elemental		Isotopic	
			% Composition (C)	(N)	Delta (C13)	(N15)
Forested OBL	OXB02	M	49.9	15.1	-32.6	4.1
		M	30.0	9.2	-35.8	4.7
		F	35.5	11.1	-35.8	4.7
		F	45.6	14.7	-36.11	4.6
	OXB06	M	45.0	13.7	-36.2	5.9
		F	48.9	13.9	-38.1	6.0
		M	53.3	12.7	-36.2	6.7
		F	47.6	14.9	-34.1	4.8
Mixed BVL	BOS10	M	46.8	14.2	-35.0	6.1
		F	49.3	13.1	-33.5	6.2
		M	53.0	15.1	-32.5	6.2
		F	50.2	13.6	-32.6	5.3
Grassed OBL	OXB05	F	38.5	11.6	-30.8	5.3
		F	49.2	13.9	-28.3	5.1
		M	49.3	13.9	-28.8	5.1
		M	36.1	10.7	-29.2	5.6
Grassed BVL	DAV01	M	47.5	14.2	-30.3	5.7
		F	44.5	12.6	-26.7	5.9
		F	48.3	14.4	-32.5	5.2
		M	45.1	12.8	-25.7	5.9
Mixed OBL	LEV01	M	23.6	6.8	-33.8	4.4
		F	49.3	11.5	-33.4	5.4
		F	48.5	12.8	-32.1	4.2
		M	55.1	10.9	-28.9	4.2

#### Appendix 5.4 Zooplankton taxa in diet

Taxa	OXB01	OXB02	OXB06	BOS10	OXB05	DAV01	LEV01
<b>Rotifera</b>							
<i>Keratella sp</i>	✓		✓		✓	✓	
<i>Brachionus sp</i>			✓	✓	✓	✓	✓
<i>Asplanchna sp</i>		✓		✓	✓	✓	✓
<i>Filinia sp</i>	✓		✓	✓	✓	✓	
<b>Cladocera</b>							
<i>Diaphanosoma sp</i>					✓	✓	✓
<b>Copepoda</b>							
<i>Mesocyclops sp</i>				✓	✓	✓	✓

**Appendix 6.1 Reproductive parameters of *N. papuensis* in all lakes**

Parameter	Sex	Statistic	Forested				Mixed BVL		Grassed OBL		Grassed BVL		Mixed OBL			
			OXB01		OXB02		OXB06		BOS10		OXB05		DAV01		LEV01	
			Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry	Wet	Dry
<b>Gonad Stage</b>	F															
	2	%	1.88	50.00	22.08	8.70	10.89	22.50	0.79	4.13	25.32	11.67	5.29	13.79	15.85	12.24
	3	%	18.78	22.22	16.88	8.70	16.83	17.50	20.63	23.14	35.44	41.67	31.75	30.17	32.93	34.69
	4	%	6.57	0.00	15.58	15.22	7.92	2.50	31.75	21.49	10.13	16.67	30.16	16.38	3.66	18.37
																32
	M															
	2	%	14.55	22.22	6.49	19.57	25.74	12.50	4.76	2.48	8.86	6.67	3.17	13.79	7.32	2.04
	3	%	56.34	5.56	3.90	19.57	13.86	30.00	5.56	9.09	5.06	8.33	3.17	6.03	17.07	4.08
	4	%	1.88	0.00	35.06	28.26	24.75	15.00	36.51	39.67	15.19	15.00	26.46	19.83	23.17	28.57
<b>Fecundity</b>	F	Mean	22688	23328	29282	24642	31378	40628	36962	56721	35021	41652	55097	126812	66760	35798
		se	5033	1702	7517	9345	5742	4153	4870	7055	6683	7145	7478	14916	22870	12676
		min	2375	18150	2375	5475	2457	14179	7698	7063	7898	10456	4540	10729	18191	18200
		max	41570	30042	70600	50269	76600	76600	64011	135067	104304	91200	268480	260123	125723	97164
<b>Oocyte diameter</b>	F	mean	0.47	0.44	0.49	0.55	0.41	0.53	0.58	0.50	0.47	0.48	0.58	0.52	0.50	0.55
		se	0.04	0.05	0.04	0.03	0.05	0.04	0.02	0.04	0.03	0.04	0.01	0.03	0.07	0.03
<b>GSI</b>	F	mean	2.85	1.31	1.67	2.77	0.93	2.48	3.50	3.22	2.87	2.41	2.73	4.38	3.01	3.56
		se	0.13	0.20	0.21	0.79	0.15	0.32	0.18	0.26	0.15	0.25	0.13	0.30	0.25	0.31
	M	mean	1.21	0.60	1.20	1.13	0.59	1.60	2.47	1.66	1.29	0.88	1.82	1.65	2.42	2.41
		se	0.06	0.11	0.15	0.13	0.08	0.18	0.21	0.13	0.56	0.21	0.14	0.16	0.21	0.41
	F+M	mean	1.92	1.02	1.47	1.58	0.72	1.90	3.01	2.44	2.55	2.01	2.44	3.16	2.76	3.20
		se	0.07	0.14	0.14	0.26	0.08	0.18	0.14	0.16	0.14	0.21	0.10	0.22	0.17	0.26
		<b>Total n</b>	<b>155</b>	<b>5</b>	<b>35</b>	<b>31</b>	<b>65</b>	<b>23</b>	<b>59</b>	<b>62</b>	<b>23</b>	<b>18</b>	<b>62</b>	<b>46</b>	<b>39</b>	<b>17</b>



**Appendix 6.2 Sex size distribution parameters for *N. papuensis***

Lake	Sex	Distribution Parameters				
		Kurtosis	Median (mm)	Mean (mm)	Mode (mm)	Skew direction
OXB01	F	2.767	180	175.83	182	-
	M	0.453	152	152.86	146	+
OXB02	F	0.621	149	152.02	163	+
	M	-0.331	130	127.11	147	-
OXB06	F	-0.695	139.5	153.27	118	+
	M	-0.617	130	136.35	108	+
BOS10	F	-0.072	201	206.85	197	+
	M	0.372	176	174.12	180	-
OXB05	F	1.004	229	217.98	231	-
	M	0.024	189	184.32	175	-
DAV01	F	0.836	215	209.87	244	-
	M	-0.710	163.5	161.58	195	-
LEV01	F	-0.387	197	160.87	176	-
	M	2.697	181.5	175.07	185	-