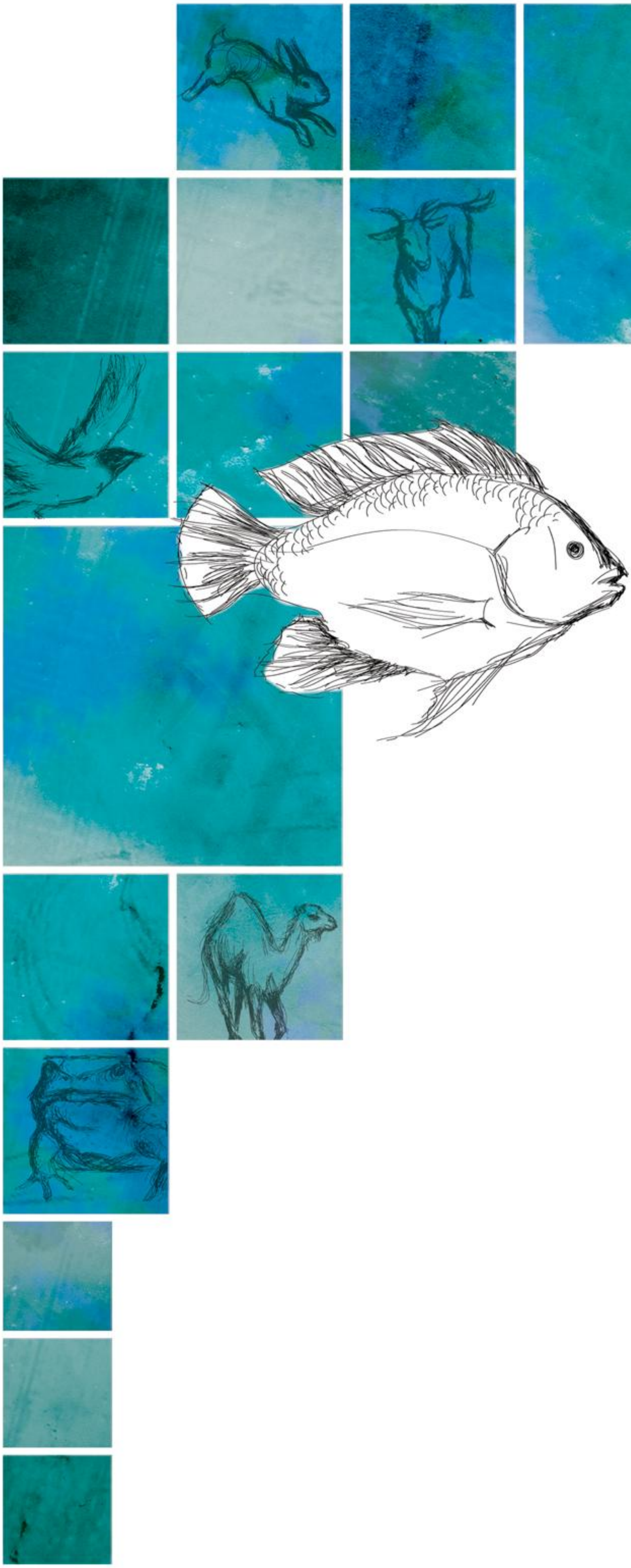
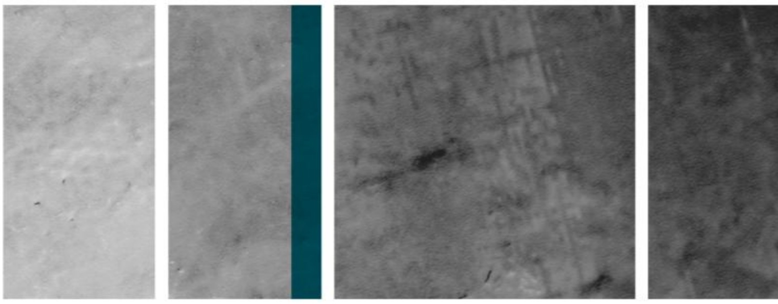


Tilapia in Australia

Development of management strategies for the control and eradication of feral tilapia populations in Australia

Russell DJ, Thuesen PA and Small FE



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Development of management strategies for the control and eradication of feral tilapia populations in Australia

John Russell
Paul Thuesen
Fiona Small

Department of Employment, Economic Development and Innovation

PO Box 5396
Cairns, Queensland 4870

2010

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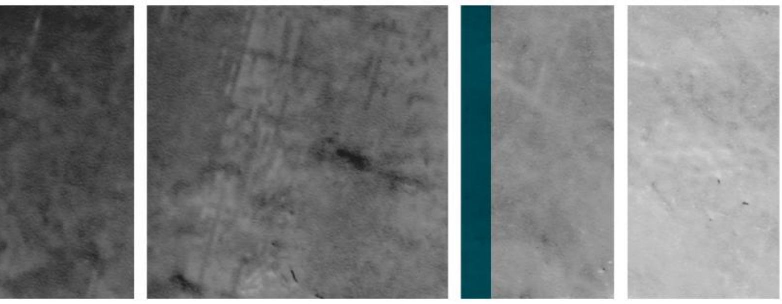

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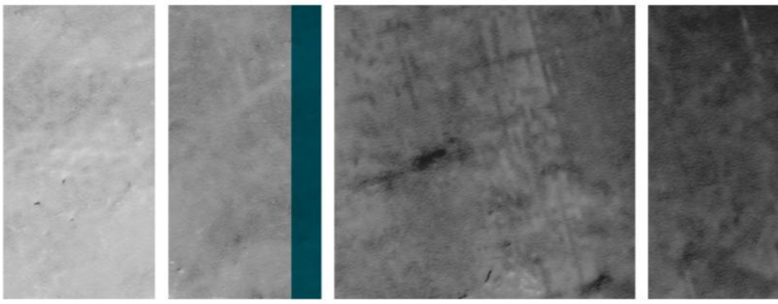
Front cover photo: Mozambique mouth brooder, *Oreochromis mossambicus*, with eggs. Image: Department of Employment, Economic Development and Innovation, Queensland



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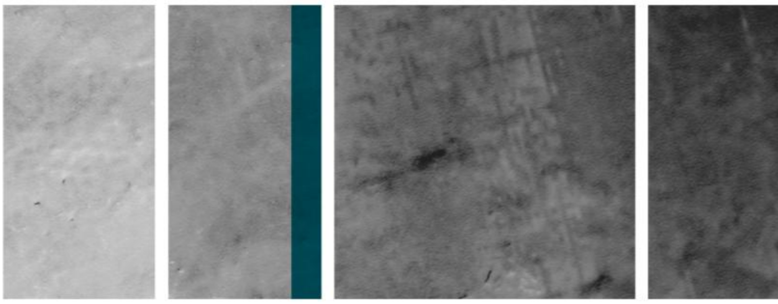


Summary

This report presents a culmination of different research projects on two species of tilapia (*Oreochromis mossambicus* (including interspecies hybrids) and *Tilapia mariae*) and provides recommendations for the future management and research of these pest fish. Feral populations of *O. mossambicus* and *T. mariae*) are now widely distributed in tropical northeastern Queensland, with *O. mossambicus* also occurring in southeastern Queensland and river systems of Western Australia. *O. mossambicus* is known to have existed in impoundments in southeastern Queensland, as well as urban drains and ornamental ponds in the Townsville region of north Queensland from about the late 1970s, while *T. mariae* became established in some eastern-flowing tropical streams by the early 1990s. In Australia, feral stocks of tilapia are widely regarded as pests that potentially threaten both native fish stocks and biodiversity.

In the first section of this report we describe a study of feral populations of both *O. mossambicus* and *T. mariae* from five locations within a 67 km radius, showing a wide diversity and plasticity in their reproductive and growth parameters. It is thought that this inherent variability is partly responsible for their capacity to quickly and efficiently invade new and sometimes marginal areas, such as the Kewarra Beach drain examined during this study. A high level of parental care ensuring that a relatively high proportion of eggs and larvae are recruited as juveniles, and the ability to spawn multiple broods over a year-round reproductive season gives tilapia a significant competitive advantage over native fishes.

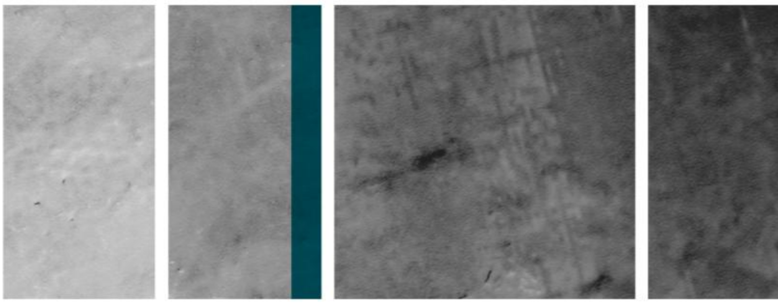
In both species of tilapia, males grew faster than females and there was evidence of considerable variability in the growth characteristic of *O. mossambicus* between study sites. In Tinaroo Falls Dam, the *O. mossambicus* population grew faster than the population in the Kewarra Beach drain, probably because the effects of harsher environmental conditions at the latter site. Ageing studies indicate that Tinaroo Falls Dam was more recently colonised by *T. mariae* than the Mulgrave River suggesting that invasion of north Queensland habitats by this species ongoing.

The CARPSIM model was successfully adapted for use with *O. mossambicus* and *T. mariae*. Some of the simulations completed suggest that it is possible to drive tilapia populations to pseudo-extinction using very high levels of fishing pressure or a combination of fishing pressure and recruitment failure. However, simulations that used more realistic estimates of fishing effort, levels of spawning disruption and other interventions, suggested that while it was possible to drive tilapia abundance down to low levels, populations would not become pseudo-extinct and would quickly recover to previous levels once interventions ceased.

An ongoing control experiment in the Herberton Weir using monthly physical removal of Mozambique tilapia via electrofishing has resulted in a substantial reduction in their relative abundance. However, fish abundance has now stabilised with the current challenge being to implement effective strategies to eliminate the remaining fish from the weir.

New tilapia invasions, both as a result of natural dispersion and through anthropogenic means, are still occurring and there is an ongoing need to continue to address this problem through targeted education programs and community engagement as well as further research. Future research is recommended in the areas of:

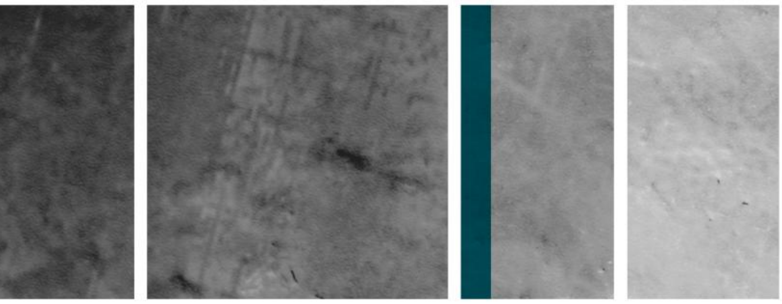
- Continuing with the control experiment at Herberton Weir
- Acoustic and radio-tagging technology
- Further investigation into the genetic population structure of *O. mossambicus* in Australia
- Further modelling
- Investigation into the use of pathogens to control tilapia populations in Australia
- Examining the feasibility of sex-biased control strategies using chemical and/or genetic technologies.



1 Introduction

Tilapia is the common name for a large number of species within the cichlid tribe Tilapiini, particularly the larger species of the genera *Tilapia*, *Sarotherodon* and *Oreochromis* that are most commonly caught in wild fisheries, or used globally in aquaculture (McAndrews 2000). The tilapiines are exclusively a pan-African assemblage of fish absent only from the northern Atlas mountains and southwest Africa (McAndrews 2000). Two species, *Tilapia mariae* and *Oreochromis mossambicus* have been introduced in Australia. *T. mariae* is endemic to the west African region, where it ranges from the Cote d'Ivoire through Ghana and Nigeria to Cameroon (King and Etim 2004). Unlike other tilapiines, it is not yet extensively cultured, probably because of inadequate information on its biology (McAndrews 2000). The other introduced species, *O. mossambicus*, is native to the eastward-flowing rivers of Central Africa. Its natural distribution extends northwards to below the Kapachera Falls in the lower Shire River of southern Malawi, including the lower Zambezi and southwards through Mozambique and all coastal rivers along the southeastern African coast to Algoa Bay (Pullin 1988).

Tilapia species are highly invasive and exist under feral conditions in every country where they have been cultured or introduced, including Australia (Canonico et al 2005). *O. mossambicus* is on the International Union for Conservation of Nature and Natural Resources list of 100 of the world's worst invasive alien species (Lowe et al 2000). Early records show that *O. mossambicus* was present in impoundments in southeastern Queensland, as well as urban drains and ornamental ponds in the Townsville region of north Queensland from about the late 1970s (Arthington et al 1984, Arthington and Milton 1986). It has since been found further north in catchments in the Cairns area, including the Atherton Tablelands (Russell et al 2003, Webb 2007) and in the Burdekin River to the south of Townsville (Webb 2007). Populations also exist in four river systems in Western Australia (Arthington and Milton 1986, Morgan et al 2004). The Australian distribution of *T. mariae* is disjunct and is currently limited to northeastern drainage catchments between latitudes 15°30' S and 18° S (Russell et al 2003, Webb 2007). *T. mariae* were resident in northeastern Queensland rivers at least as early as 1991 (Russell and Hales 1993), although a discreet population existed in a power station cooling pond in Victoria much earlier (Cadwallader et al 1980). Recently a small population of *T. mariae* was found in the western-flowing Gulf of Carpentaria drainage and apparently successfully eradicated in late 2008 using a piscicide (M Pearce, Queensland Department of Employment, Economic Development and Innovation, unpublished data).



1.1 Project objectives

This current study was developed for the Invasive Animals Cooperative Research Centre (IA CRC) to address its operational target of ‘reducing the spread and impacts of pest fish leading to improved water resource management’. The specific objectives of the study were:

1) Collection of critical ecological and life history data to enable implementation of successful control strategies and assist in the planning of education and extension programs within the Queensland Department of Employment, Economic Development and Innovation.
2) Modelling and evaluation of a range of management scenarios that could be used for the control of tilapia in Australia.
3) Recommending and participating in the development of control strategies for tilapia in Queensland.

In addition, a comprehensive literature review of *O. mossambicus* including its distribution, impacts, control strategies and methodologies, trophic ecology and ecological and environmental tolerances was published in the peer-reviewed journal *Reviews in Fish Biology and Fisheries* (Russell et al, 2012). This complements a similar document on *T. mariae* that was prepared by the CSIRO Sustainable Ecosystems group (Bradford et al, 2011). The Principal Investigator is a co-author of this review.

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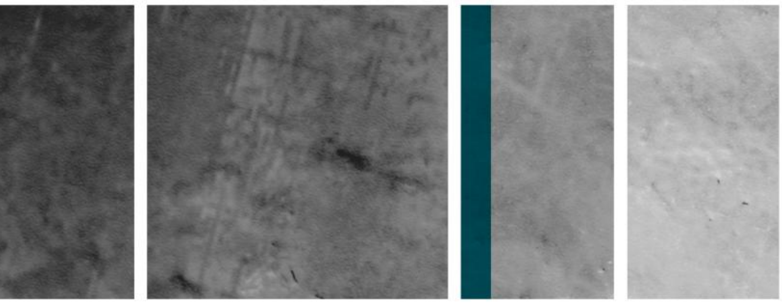


2 General methods

2.1 Study sites

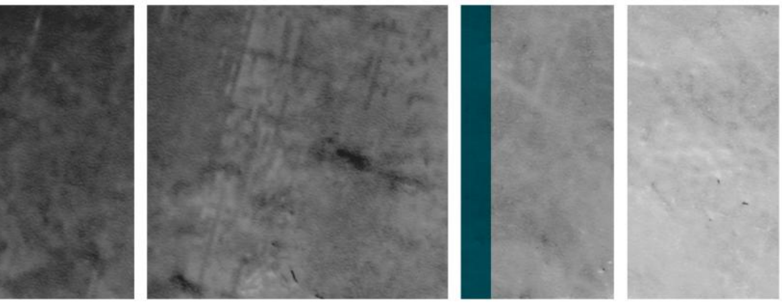
During this study a number of sites throughout northern Australia were sampled for both species of introduced tilapia and these are shown in Figure 1. The most northern study site consisted of a series of four shallow, eutrophic freshwater ornamental ponds on the Paradise Palms golf course (16°47’ S, 145°26’ E). These ornamental ponds, which have a total surface area of <2 ha and a maximum depth of about 2 m, receive treated sewerage effluent that is used to irrigate the golf course. At nearby Kewarra Beach, a 100-m section of a highly modified, shallow drain was also sampled. The upper section of this drain is concrete lined and has a maximum width and depth of about 3 m and 0.4 m respectively. The lower section of the drain at the time of sampling was choked with exotic grasses (predominantly *Urochloa mutica*) and had a width of between about 2 m and 4 m and depth averaging between approximately 0.3 m and 1.5 m. During the dry winter and spring months, the drain generally stopped flowing and was reduced to a series of isolated pools. A third study site was in the lower, freshwater reaches of the Mulgrave River (17°24’ S, 145°54’ E). This site included the main river channel and adjacent lagoons and watercourses and was close to the upper limit of tidal influence.

Tilapia were also sampled at two inland locations: Tinaroo Falls Dam and the adjacent Herberton Weirs. Tinaroo Falls Dam (17°09’ S, 145°32’ E) is a relatively large irrigation impoundment situated on the upper reaches of the Barron River at an altitude of 670 m. It has a storage capacity of 436.5 GL, a surface area of 33.7 km^2 and a maximum depth of 41.8 m at full storage level (FSL). The Herberton Weirs (17°22’ S, 145°25’ E) are constructed at an altitude of about 1020 m on a tributary of the upper Herbert River and provide potable water for the nearby township of Herberton. The two weirs are approximately 800 m apart, with the upper weir having a surface area of around 7.6 ha and a maximum depth at FSL of 10 m. The lower weir is older and has a surface area of about 2 ha and a maximum depth at FSL of 8.3 m. Hobo® pendant temperature and light data loggers, programmed to log surface and bottom temperatures at hourly intervals, were set at the Herberton top weir in January 2007. Hourly surface-water temperatures for the same period for the Mulgrave River were obtained from Queensland Department of Environment and Resource Management data loggers, but only opportunistic temperature data for Tinaroo Falls Dam were obtained from the water provider (Sunwater Corporation). No temperature data were was available for the other study sites.

2.2 Fish Sampling

Fish were sampled monthly at each of the study sites (Figure 1) using a range of techniques. These included a boat-mounted electrofisher (Mulgrave River, Herberton weirs and Tinaroo Falls dam), a backpack electrofisher (Kewarra Beach Drain) and cast netting (Paradise Palms). The 4.3 m electrofishing boat was equipped with a Smith-Root® Model 7.5 Generator Powered Pulsator (Vancouver, WA) and was used to generate a pulsed DC waveform with voltages ranging from 135 to 1000 V; where possible, the higher voltages were used. In the lower, tidally-influenced Mulgrave River, the conductivity at the site was measured to determine the most efficient operational setting. During electrofishing the vessel was positioned roughly parallel to and about 5 m out from the bank or navigated up the centre of narrower watercourses. The boat was manoeuvred in such a way that the anode array was nearly always in close proximity to suitable fish habitat such as snags, grasses, macrophytes, overhangs and rocky structures. All electrofished tilapia were collected with dip nets with 3 m long handles, and immediately placed into a 100 litre recirculating live tank onboard the vessel. At the completion of sampling at each site, or when the live tank was determined to be approaching maximum capacity, all fish were euthanased with an overdose of the anaesthetic Aqui-S (Aqui-S New Zealand Ltd.) and immediately chilled with ice before being returned to the laboratory for further processing. In shallower water, a 600 V Kainga EFM300 backpack electrofisher (NIWA Instrument Systems, New Zealand) and cast or throw net (2.5 m net, 20 mm SM, monofilament or braided monofilament; 80% extension or greater) were used for sample collection. All fish collected using these methods were euthanased and processed as described above.

At each of the study sites, a minimum of 50 fish were caught, with initial preference given to larger fish over juveniles to maximise the number of mature fish sampled for reproductive studies. From approximately mid 2007, a shortage of larger fish at most sites prompted a change in sampling strategy to include all fish regardless of size. The laboratory methods used to process these fish and data analysis techniques are detailed in later chapters.



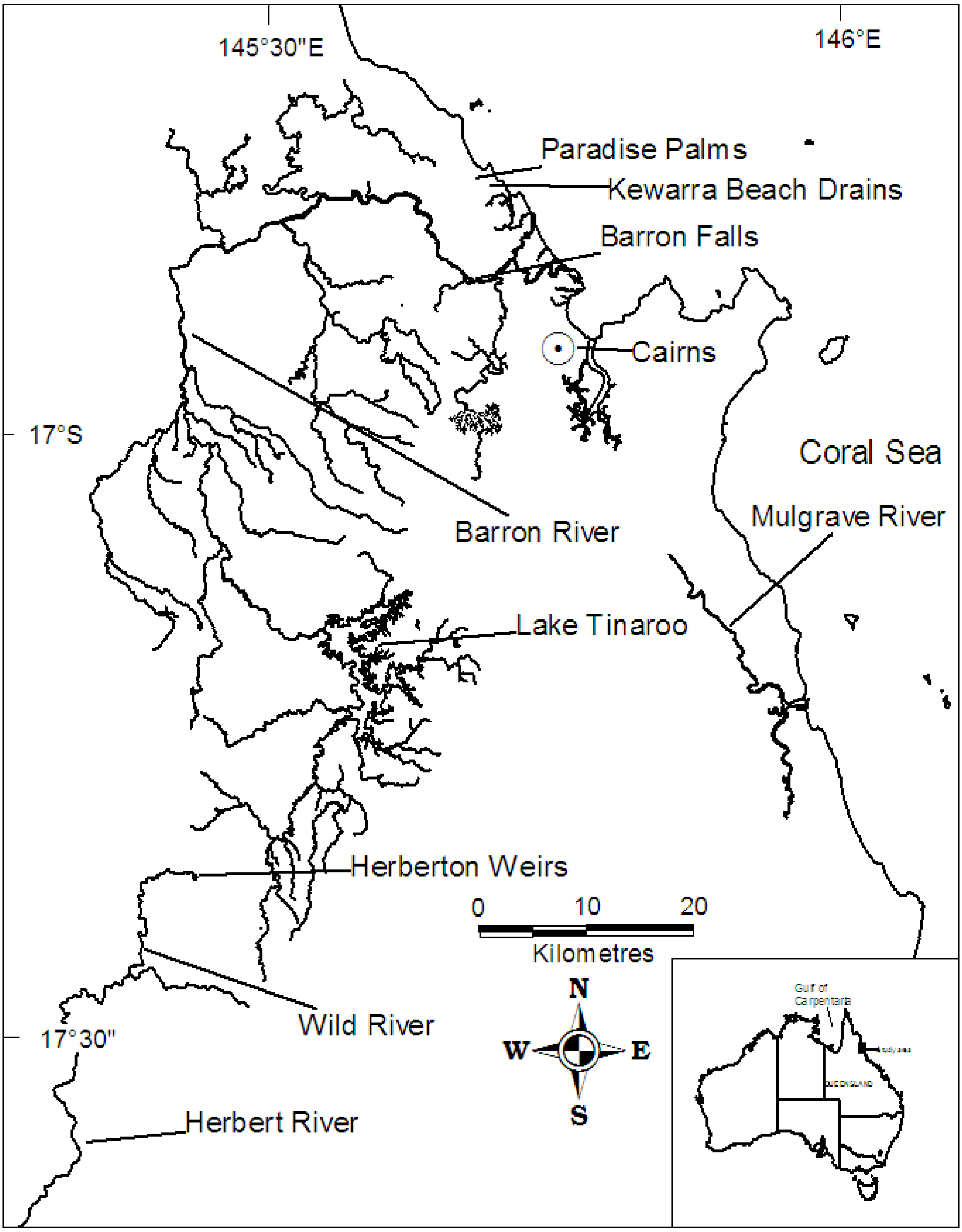
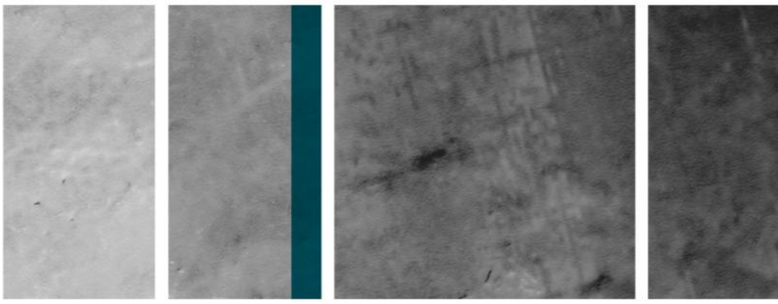


Figure 1: Study locations in Northern Australia



3 Reproductive strategies

3.1 Introduction

The life history traits and reproductive strategies employed by tilapiine fishes are part of the reason that they are successful invaders. These traits, including parental care, a protracted breeding season that is continuous in some areas, iteroparity in spawning, a relatively large egg size, the capacity to mature at an early age and ‘stunting’, combine to ensure a generally high production and survival of eggs and larvae (Arthington and Milton 1986, King and Etim 2004, Canonico et al 2005). In some species (eg *T. niloticus*), parental care of young may continue for extended periods of up to three weeks (Peterson et al 2004). However, this relatively large investment of energy in parental care does come at a cost. Jiménez-Badillo (2006) argued that in *O. aureus* the limited amount of feeding opportunities that occur during incubation and brood guarding can contribute to reductions in growth rate, which can occur over an extended period. A strategy adopted by some species to overcome this cost is for larger females to maximise their lifetime reproductive success by producing young all year round but at a lower brooding efficiency (Courtenay and Hensley 1979). Alternatively, dependent upon environmental conditions, some tilapiine species might invest in only one brood per year (Reardon and Chapman 2008). This apparent plasticity in reproductive ability has contributed to the success of tilapiine fishes globally.

A number of environmental cues have been suggested as triggers for reproductive activity in cichlids. These include rainfall, food availability, photoperiod and lunar synchronicity (Reardon and Chapman 2008). Okorie (1973) noted that in *T. niloticus*, there was a higher proportion of spawners of both sexes present during the full moon, whereas Schwanck (1987) found that in *T. mariae*, most egg clutches were laid during the last quarter of the moon cycle. Schwanck (1987) suggested that spawning before the full moon enhanced the effectiveness of parental care by allowing a maximum amount of moonlight during the most critical phases of larval development, and also provided a spawning cue for pairs. Reardon and Chapman (2008) also noted a correlation between gonad mass and total rainfall for the preceding month, suggesting that rainfall, or a correlate thereof, might be a trigger for inducing gonad maturity.

Many species of tilapia, particularly those resident in tropical areas, have an extended breeding season often covering most of the year (Webb 1994, Ikomi and Jessa 2003), which peaks during the rainy season (Peterson et al 2004). For example, in *T. mariae*, Ikomi and Jessa (2003) found that spawning occurred throughout the year with a peak at the beginning of the wet season, with another minor peak at the height of the floods. In *O. aureus*, where ovarian maturation is asynchronous, egg liberation could take place every 45 days over a period of four months (Jiménez-Badillo 2006). In addition, maternal mouth-brooders like *O. mossambicus* do not have strict habitat requirements for reproduction, so can spawn in all available habitats or colonise new environments by transporting young in their mouths (Canonico et al 2005). This highly complex pattern of behaviour has allowed tilapiine species to successfully invade communities that would otherwise be dominated by uniparental fish species (Annett et al 1999).

This chapter gives the results of a two-and-a-half year study of the reproductive strategies and biology of populations of *T. mariae* and *O. mossambicus* (including interspecies hybrids) resident in various freshwater habitat types, including coastal rivers, impoundments, ornamental ponds and urban drains. This work was part of a broader study investigating options for the control and management of feral populations of these species in northern Australia.

3.2 Methods

3.2.1 Fish sampling

Details of the study areas and fish sampling techniques are given in the General Methods section.

3.2.2 Laboratory analyses

In the laboratory, all fish were first measured to the nearest 1 mm and weighed to the nearest 1 g before being dissected to determine their sex and to assess their reproductive maturity.

The gonads were carefully excised, trimmed to remove extraneous fat and connective tissue and then weighed (to the nearest 0.1 g). The gonad development was assessed macroscopically by assigning the gonads an index of maturity using a slightly modified version of the six-point gonad-maturity classification scheme described by Davis (1982). The stages used were: 1 — immature; 2a — developing (virgin); 2b — developing; 2c — recovering; 3 — maturing; 4 — mature; 5 — ripe; and 6 — spent. To validate these macroscopic assessments, a sample of approximately 500 gonads was preserved in 10% neutral buffered formalin for subsequent sectioning, staining and histological examination.

The gonads were processed for histological examination by first taking medial sections (4 μm) that were stained regressively using Harris's haematoxylin and water-soluble eosin before mounting in Depex® (Gurr 1973). Stained gonad sections were examined microscopically and assigned a maturity stage using criteria similar to those described by (Davis 1982). Mean monthly gonadosomatic indices (GSIs) were used to determine maturity and reproductive seasonality. GSIs (gonad weight/total body weight-gonad weight × 100) were calculated monthly for mature fish (ie stage 2b gonads or later stages) and data were pooled where the same month was sampled in different years. Potential fecundity (PF), which is the total number of hydrated oocytes, or oocytes undergoing final maturation in each gonad that can potentially be released, was calculated for each mature fish. Relative fecundity (RF), which is a measure of potential fecundity per unit of body weight, was obtained by dividing fecundity by total body weight. Fecundities were log-transformed prior to analyses using ANOVA and a post hoc least significant difference test.

The cumulative percentage of developing/mature/ripe/spent (stages 2b–6) fish in each 25 mm size class or year class was determined for both male and female fish at all sites and then regressed (y) against size class (x) using the three parameter logistic function:

$$y = \frac{a}{1 + \left(\frac{x}{x_0} \right)^b}$$

where X_0, a and b are constants with a = 100 (Zar 1984). The age- and length-at-50%-maturity (Lm_{50}) was estimated from the point on the curve where 50% of fish were mature. Chi-square tests were used to compare combined sex ratio data for 25 mm size classes (Zar 1984). Statistical comparisons were made using Genstat Release 11.1 (VSN International, Oxford, United Kingdom), and Sigmaplot v8 (Systat Software, San Jose, California, United States) was used for determining the logistic functions.

3.3 Results

3.3.1 Length distributions

Length–frequency distributions of sexed male and female tilapia caught at all study sites are shown in Figure 2 and Figure 3. Males dominated the larger size classes for both *T. mariae* and *O. mossambicus* at all sites sampled during the study. There was a significant difference ($P<0.01$) between the mean lengths of *O. mossambicus* caught at the study sites; individual sites were all significantly different (LSD, $P<0.05$) from each other. The lengths of all *T. mariae* caught in the Mulgrave River were significantly greater ($t = 7.93$, d.f. = 1588, $P<0.01$) than specimens caught in Tinaroo Falls Dam. The size of male *T. mariae* from Tinaroo Falls Dam ($t = 10.64$, d.f. = 627, $P<0.01$) and the Mulgrave River ($t = 19.61$, d.f. = 740, $P<0.01$) was significantly greater than the size of females. Similarly, *O. mossambicus* males were significantly larger than females caught at Herberton Weir ($t = 10.24$, d.f. = 778, $P<0.01$), Kewarra Beach Drains ($t = 3.71$, d.f. = 525, $P<0.01$), Paradise Palms ($t = 9.23$, d.f. = 642, $P<0.01$) and Tinaroo Falls Dam ($t = 11.41$, d.f. = 296, $P<0.01$).

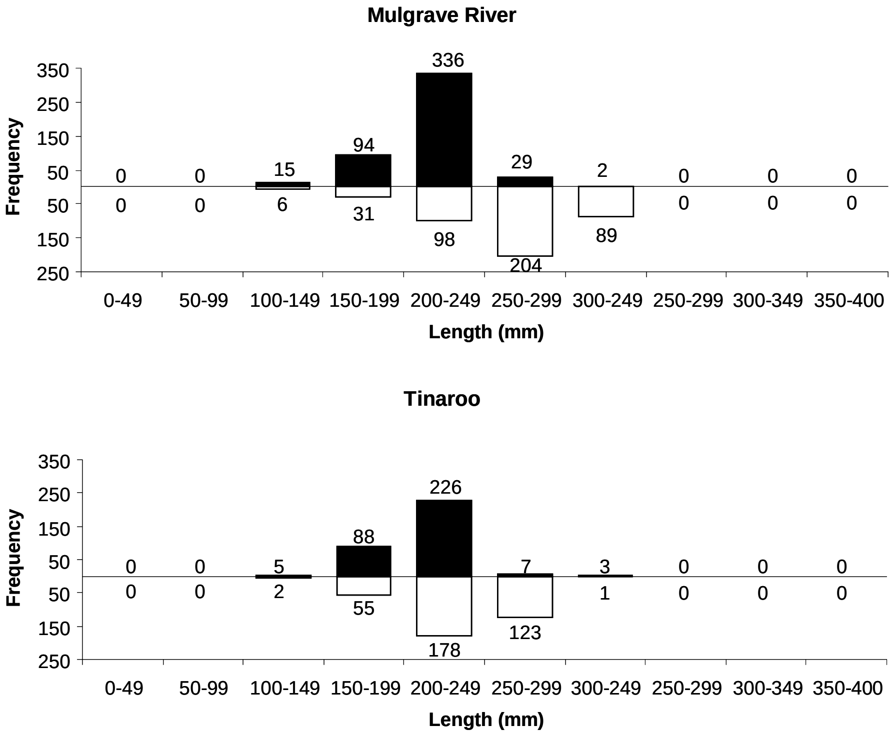


Figure 2: Length–frequency distributions of *T. mariae* males (open bars) and females (solid bars).

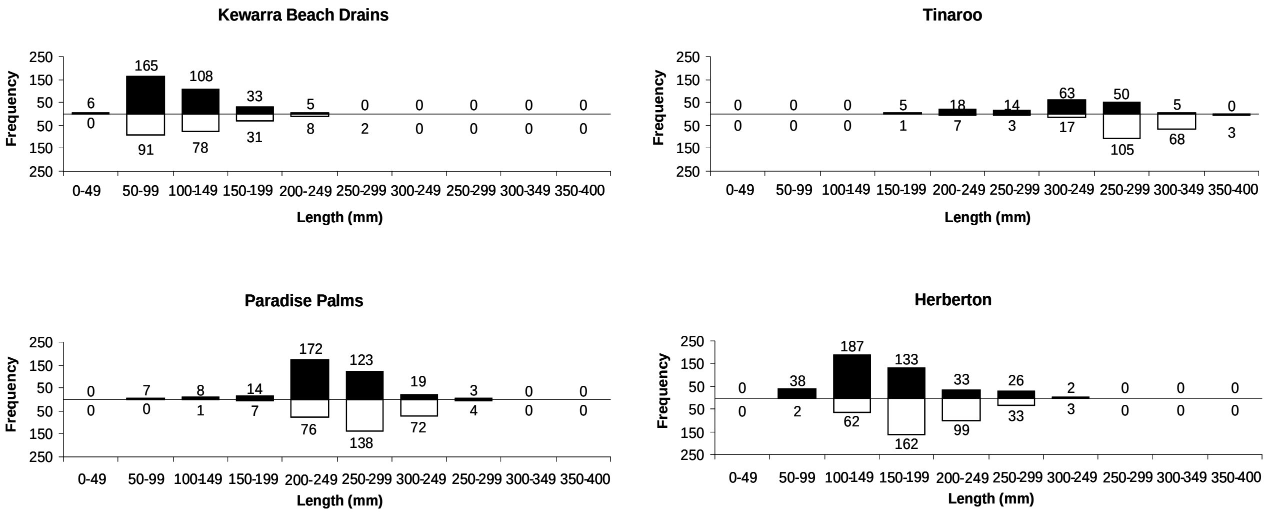
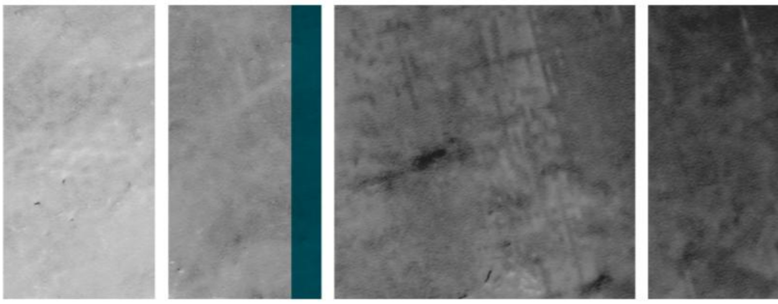


Figure 3: Length–frequency distributions of *O. mossambicus* males (open bars) and females (solid bars).

3.3.2 Reproductive seasonality

The seasonal changes in GSI for female *T. mariae* and *O. mossambicus* at all sampling locations are shown in Figure 4 and Figure 5 respectively. At the coastal river location, average monthly GSIs for *T. mariae* were relatively low, varying between 0.7 and 1.34 with no large seasonal peaks. There was no significant difference between the average GSI values between October 2006 and December 2007, suggesting relative stability. The gonads of fish sampled all year round had developing and mature oocytes present, but spawning (stage 5) and spent fish (stage 6) were not sampled from May to July inclusive (Figure 6). In contrast, the GSIs for the *T. mariae* population in Tinaroo Falls Dam were much more variable, with average monthly values peaking at 3.03 in February before dropping to a low of 0.44 in May. Spawning (stage 5) and/or spent fish (stage 6) were present most months from March to May inclusive. Resting fish (stage 2b) made up more than 75% of all the mature fish sampled (Figure 6). Even though water temperatures were still relatively low, the GSIs increased from June to August before stabilising during the Australian spring and summer months.

The seasonal changes in GSIs for *O. mossambicus* sampled at all sites are shown in Figure 5 and the monthly percent frequency of gonad stages are shown in Figure 7. Average monthly GSIs at the Paradise Palms site peaked in October 2006 and October 2007 and were relatively stable for the rest of the year, generally with average values of less than one. There was no significant difference between the average GSI values between October 2006 and December 2007, and spawning (stage 5) and spent fish (stage 6) were present in most months except July and August. At the nearby Kewarra Beach Drain site, GSIs were, with the exception of a small sample taken in February 2006, relatively stable between October 2006 and September 2007 before peaking in November 2007. The average monthly GSI values at this location were generally much higher here than at any of the other sites but spawning and spent fish were only present in the months of October through January (Figure 7). The average monthly GSI values for *O. mossambicus* sampled from the Herberton Weir sites showed a marked seasonal pattern, with peak values of 2.89 in October 2006 and 3.32 in November 2007. Average GSIs remained low (<0.45) through the colder months from April through August before gradually increasing. For this period, there was little evidence of spawning activity, with mature resting



female fish dominating the sample. In September, the proportions of stage 3 and stage 4 fish increased, with spawning (stage 5) and spent (stage 6) fish present from October through to March. This seasonal trend approximately reflects the drop in water temperature shown in Figure 5. The average monthly GSI values for *O. mossambicus* in the other uplands location, Tinaroo Falls Dam, also showed a seasonal pattern, with the highest average values occurring during November (1.03 in 2006 and 0.55 in 2007) and December (0.40 in 2006 and 0.43 in 2007). Compared with the other sampled *O. mossambicus* populations, monthly GSIs at Tinaroo Falls Dam were all <1, with the exception of November 2006, which had a GSI of 1.03. For the cooler months from April to June, average monthly GSIs at Tinaroo Falls Dam were less than 0.1. The monthly percent frequencies of the various gonad stages showed that spawning (stage 5) and spent (stage 6) *O. mossambicus* were only present at Tinaroo Falls Dam in March, with most other months dominated by fish of development stages 2 and 3 (Figure 7).

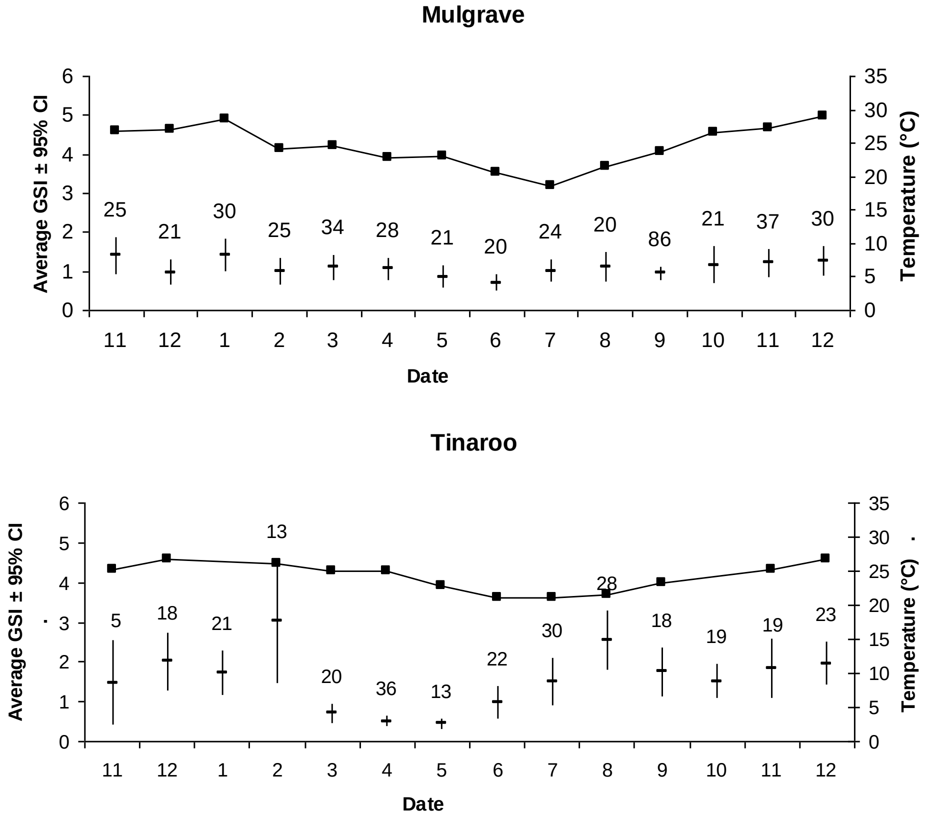
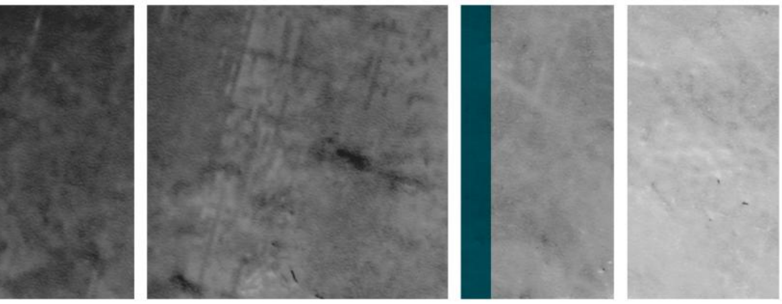


Figure 4: Average monthly GSI (± 95% Confidence Interval) for *T. mariae* sampled from November 2006–December 2007. The numbers above each monthly GSI indicate sample size. Mean monthly water temperatures (°C) are also shown.



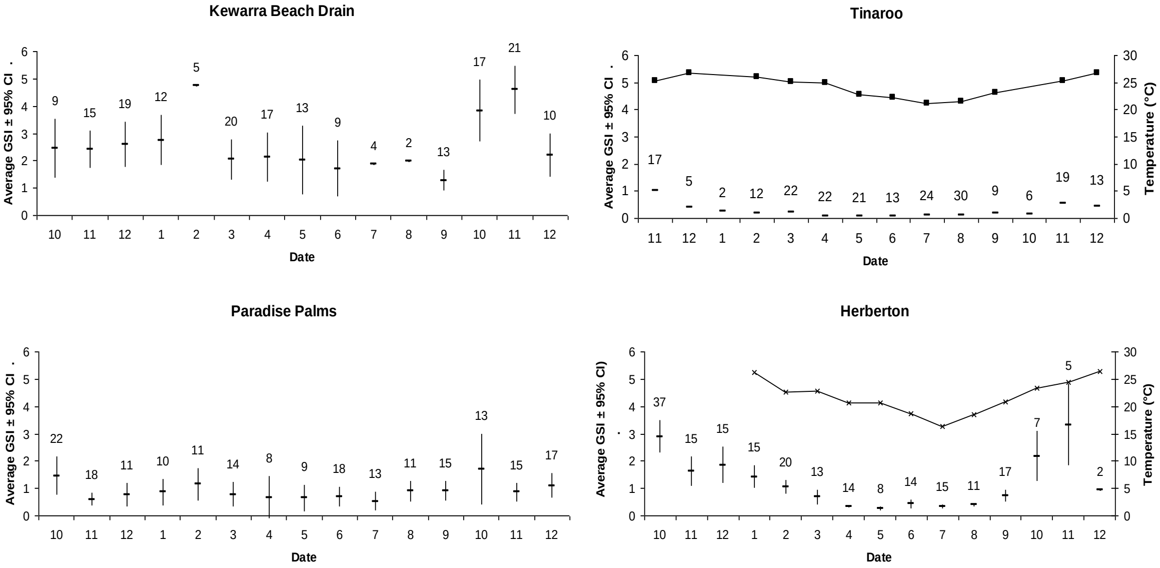
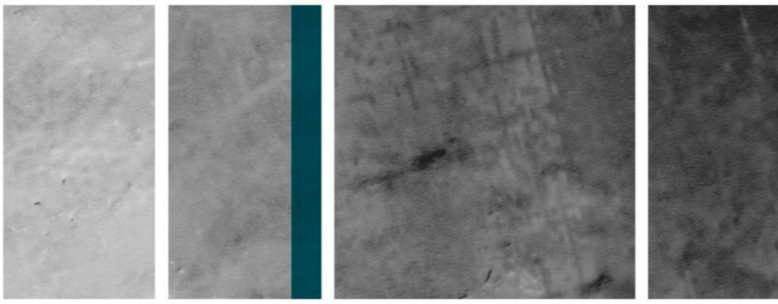


Figure 5: Average monthly GSI (± 95% Confidence Interval) for *O. mossambicus* sampled from November 2006–December 2007. Mean monthly water temperature (°C) data is shown for the Tinaroo Falls Dam and Herberton Weir sites.



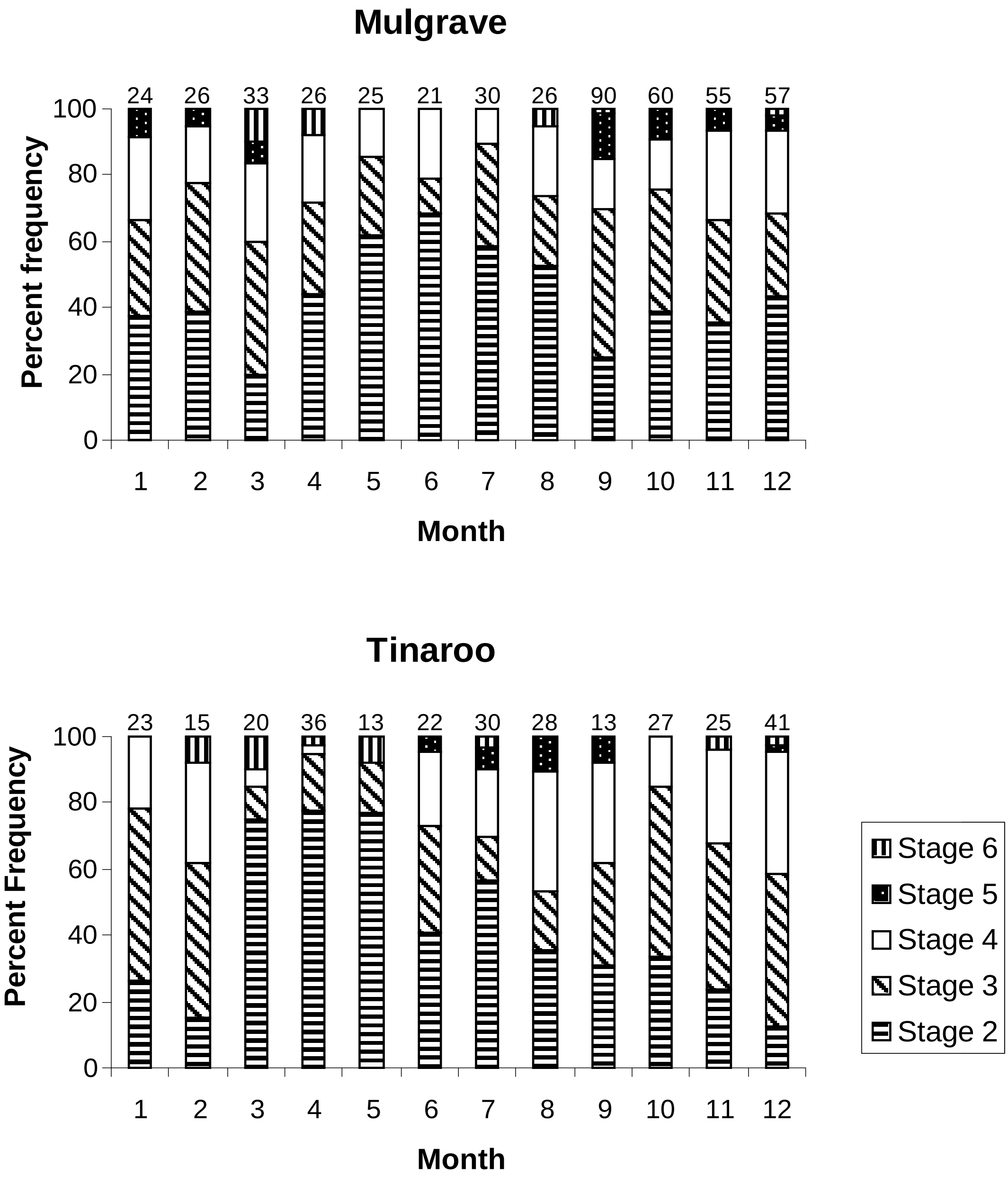
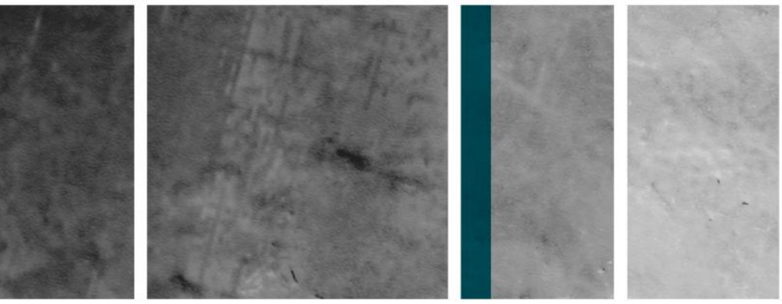


Figure 6: Monthly percentages of female *T. mariae* at different gonadal development stages from the Tinaroo and Mulgrave sites (immature fish of stages 1 and 2a are excluded from this analysis).



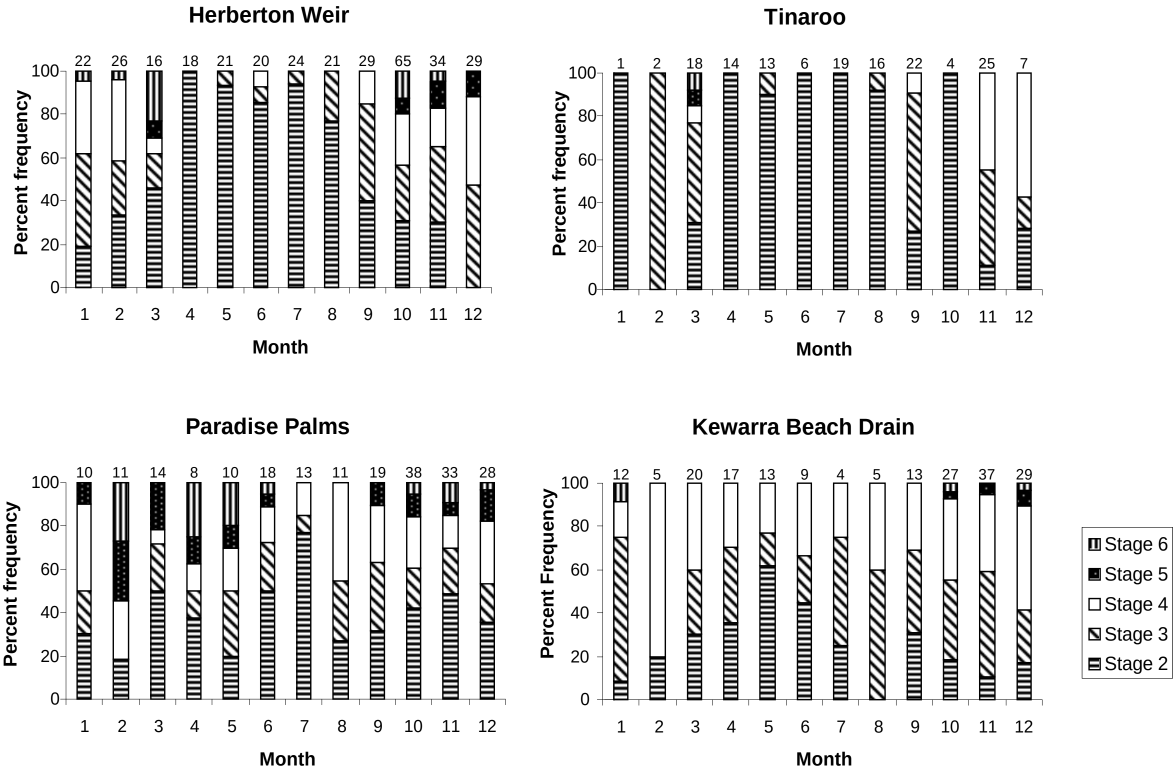


Figure 7: Monthly percentages of female *O. mossambicus* at different gonadal development stages from four north Queensland study sites (immature fish of stages 1 and 2a are excluded from this analysis).

3.3.3 Fecundity

The average total fecundity and average eggs/kg for *O. mossambicus* and *T. mariae* at the five study sites is shown in Table 1. There was no significant difference (P = 0.51) in the lengths of the stage 4 or 5 female *T. mariae* selected for fecundity estimates. There was also no significant difference between the log-transformed mean number of eggs/kg (P = 0.27) and the log-transformed total egg count (P = 0.38) for *T. mariae* caught at the Mulgrave River and Tinaroo Falls Dam sites. There was, however, a significant difference between sites in the length of *O. mossambicus* selected for fecundity estimates (P<0.01) and between both the log-transformed mean number of eggs/kg (P<0.01) and the log-transformed total egg count (P<0.01). While the log-transformed mean numbers of eggs/kg for the Paradise Palms and Tinaroo Falls Dam sites were not significantly different (P>0.05), the Kewarra Beach Drain and Herberton Weir sites were significantly less (P<0.05) than all other locations (Table 1). However, the actual average eggs/kg of 24,238 for *O. mossambicus* from the Kewarra Beach Drain, was nearly three times the nearest value (ie Herberton Weir, 8651).

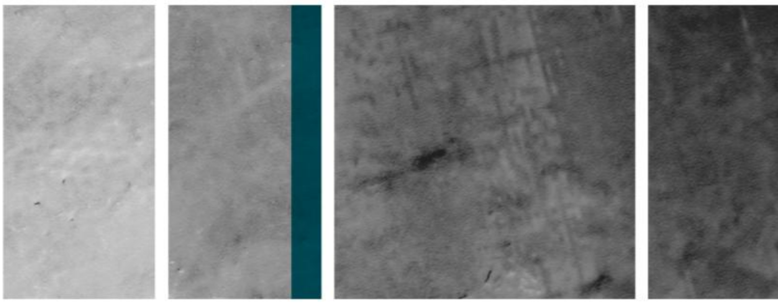


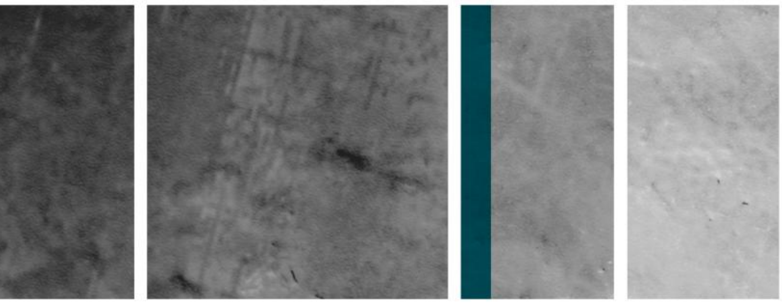
Table 1: Fecundity of *O. mossambicus* and *T. mariae* at sampled locations.

	O. mossambicus			***T. mariae***		
Location	***Number***	***Eggs/kg***	***Total eggs***	***Number***	***Eggs/kg***	***Total eggs***
Paradise Palms	29	7.95a	7.00a	ns	ns	ns
Herberton	17	8.94b	6.76a	ns	ns	ns
Kewarra Drains	26	9.82c	5.77b	ns	ns	ns
Tinaroo	13	7.74a	7.65c	16	9.33a	8.14a
Mulgrave River	ns	ns	ns	16	9.21a	8.01a

Note: Different letters in the same column indicate a significant difference. 'ns' is not sampled and all fecundity data are log transformed.

3.3.4 Length at maturity

The average length (± 95% CI) of mature *O. mossambicus* females sampled at the Kewarra Beach Drain, Herberton Weir, Paradise Palms and Tinaroo Falls Dam sites was 111.68 mm (minimum 106.1 mm, maximum 117.3 mm), 177.5 mm (minimum 171.3 mm, maximum 183.7 mm), 258.5 mm (minimum 253.9 mm, maximum 263.1 mm) and 341.6 mm (minimum 335.2 mm, maximum 348.1 mm) respectively (Figure 9). The average length (± 95% CI) of mature *T. mariae* females collected from the Mulgrave River and Tinaroo Falls Dam was 221.8 mm (minimum 219.7 mm, maximum 221.8 mm) and 213.3 mm (minimum 211.0 mm, maximum 215.6 mm) respectively (Figure 8). Male *O. mossambicus* and *T. mariae* matured at a larger size than females. The length at which 50% of female *T. mariae* sampled from Tinaroo Falls Dam and in the Mulgrave River were mature (Lm_{50}) were similar, being 212.8 mm and 220.5 mm respectively. The Lm_{50} values of male *T. mariae* from Tinaroo Falls Dam and the Mulgrave River were 245.3 mm and 271.3 mm respectively (Figure 8). Logistic curves for male and female *O. mossambicus* populations sampled during this study are shown in Figure 9. The Lm_{50} values for male *O. mossambicus* from Tinaroo Falls Dam, Paradise Palms, Herberton Weir and the Kewarra Beach Drain were 380.7 mm, 278.1 mm, 188.9 mm and 113.8 mm respectively and for females were 331.0 mm, 244.0 mm, 155.5 mm and 93.3 mm respectively. The parameters for all logistic equations are given in Table 2.



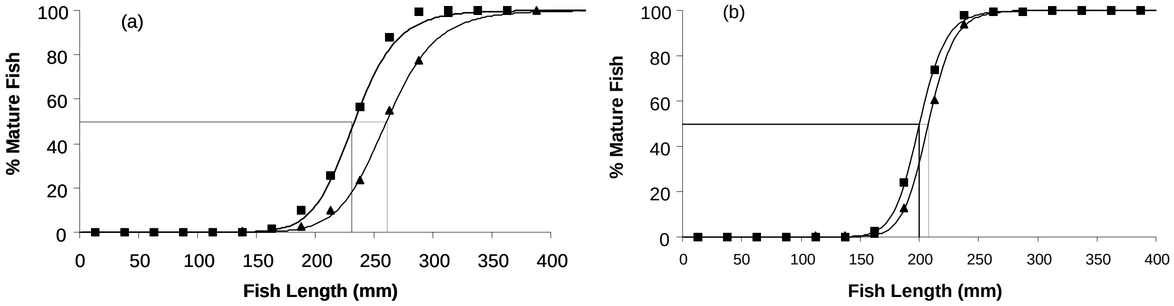


Figure 8: Length-maturity logistic curves for (a) male and (b) female *T. mariae* from the Mulgrave River (▲) and Tinaroo (■). Logistic equation parameters are given in Table 2. The vertical lines show the Lm_{50} values.

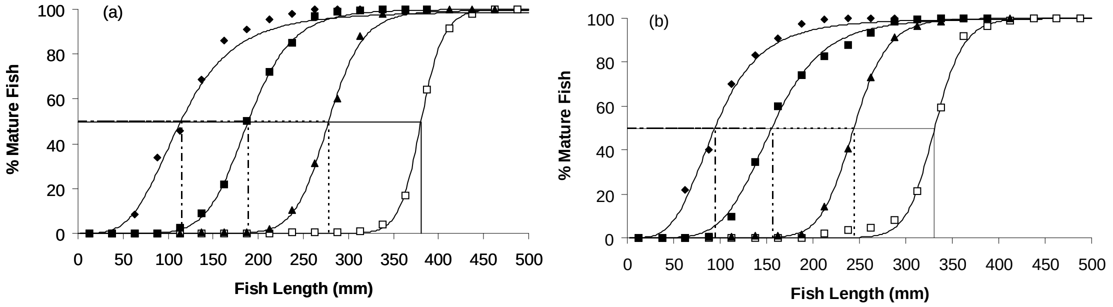


Figure 9: Length-maturity logistic curves for (a) male and (b) female *O. mossambicus* from the Kewarra Beach Drain (♦), Herberton (■), Paradise Palms (▲) and Tinaroo (□). Logistic equation parameters are given in Table 2. The vertical lines show the Lm_{50} values.

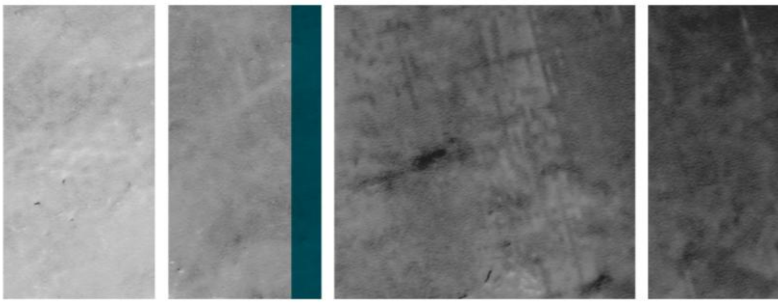


Table 2: Length parameters for logistic equations used for calculation of Lm_{50} values.

	O. mossambicus				*T. mariae*	
Length	***Paradise Palms***	***Kewarra Drains***	***Herberton***	***Tinaroo***	***Tinaroo***	***Mulgrave River***
b (males)	-14.8	-3.8	-7.95a	-31.2	-12.9	-13.6
b (females)	-13.6	-3.84	-5.69	-21.9	-18.8	-20.2
X_0 (males)	278	113	189	381	245	271
X_0 (females)	244	94.3	156	331	213	220

Note: Parameter ‘a’ was set to 100 for all calculations.

3.3.5 Age at maturity

The age at which 50% of *T. mariae* sampled from Tinaroo Falls Dam and the Mulgrave River were mature (A_{50}) was for males 1.2 years and 3.1 years respectively, and for females 1.2 years and 2.4 years respectively (Figure 10). The age–maturity logistic curves for male and female *O. mossambicus* populations sampled during this study are shown in Figure 11. The A_{50} values for *O. mossambicus* from Tinaroo Falls Dam, Paradise Palms, Herberton Weir and the Kewarra Beach Drain are for males 2.6 years, 2.2 years, 1.4 years and 0.5 years respectively and for females 2.4 years, 2.0 years, 1.6 years and 0.5 years respectively. The parameters for all logistic equations are given in Table 3.

Table 3: Age parameters for logistic equations used for calculation of A_{50} values.

	O. mossambicus				*T. mariae*	
Age	***Paradise Palms***	***Kewarra Drains***	***Herberton***	***Tinaroo***	***Tinaroo***	***Mulgrave River***
b (males)	-3.92	-1.50	-4.3	-6.76	-2.17	-3.14
b (females)	-3.46	-1.81	-3.72	-5.67	-1.96	-2.87
X_0 (males)	2.17	0.45	1.39	2.59	1.26	3.28
X_0 (females)	1.98	0.54	1.55	2.43	1.21	2.46

Note: Parameter ‘a’ was set to 100 for all calculations.

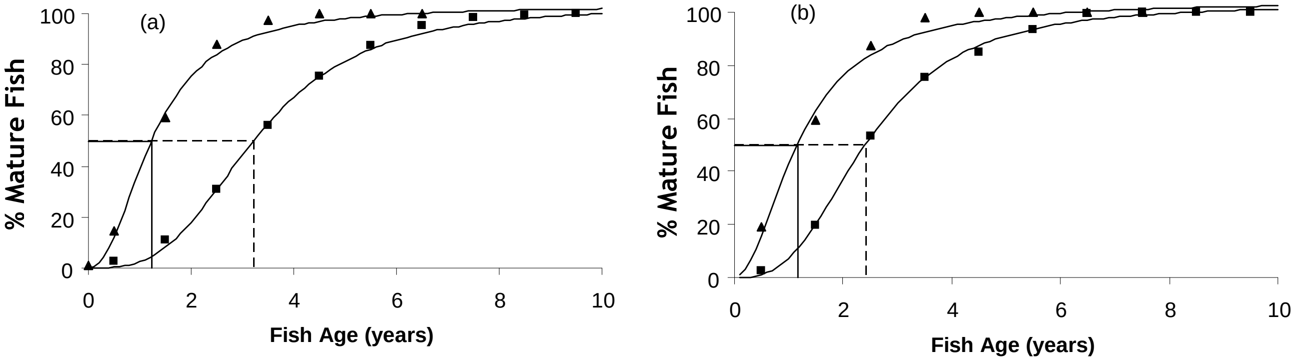


Figure 10: Age-maturity logistic curves for (a) male and (b) female *T. mariae* from the Mulgrave River (▲) and Tinaroo (■). Logistic equation parameters are given in Table 3. The vertical lines show the A_{50} values.

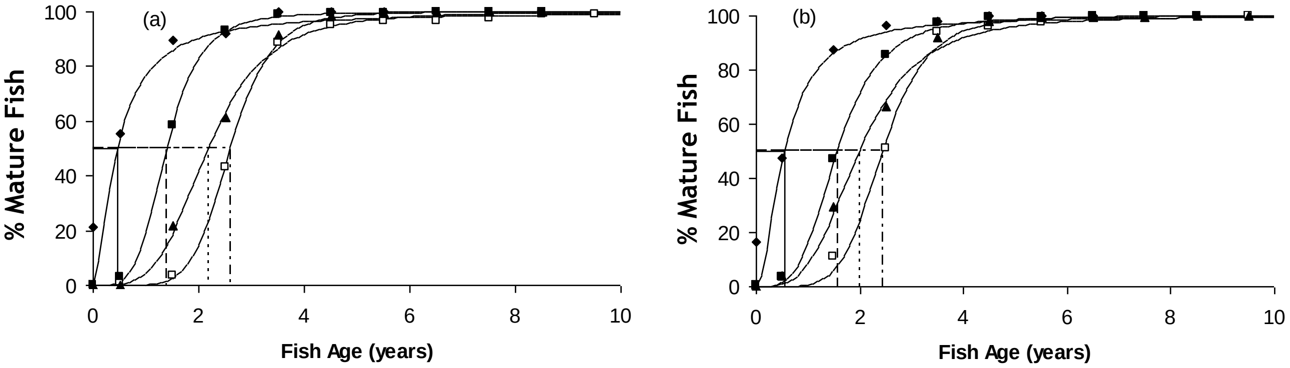


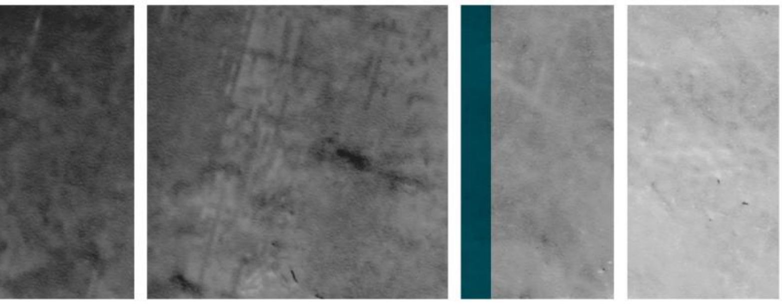
Figure 11: Age-maturity logistic curves for (a) male and (b) female *O. mossambicus* from the Kewarra Beach Drain (♦), Herberton (■), Paradise Palms (▲) and Tinaroo (□). Logistic equation parameters are given in Table 3. The vertical lines show the A_{50} values for each stock.

3.4 Discussion

The life history traits of some tilapias, including *O. mossambicus*, have been described as being somewhere on a continuum between altricial and precocial styles (Balon 1979, Noakes and Balon 1982, Arthington and Milton 1986, De Silva and Amarasinghe 1989, James and Bruton 1992). Altricial life-history styles are characterised by short growth intervals, early maturation, high fecundity, reduced parental care and short life spans, whereas precocial life-history styles are characterised by prolonged somatic growth and deferred maturation, reduced fecundity, increased parental care and extended life spans (Noakes and Balon 1982, Balon and Bruton 1986). Noakes and Balon (1982) suggest that the ability of some tilapiine species to shift along the continuum from precocial to altricial life-history styles, is a reason why some tilapias are extremely successful at invading newly created marginal habitats. There is evidence to support this phenomenon from tilapia populations that have established in other parts of Australia (Arthington and Milton 1986) and from the populations examined in this study. Arthington and Milton (1986) argued that both of the populations of *O. mossambicus* that they investigated in southeastern Queensland conformed to the precocial life-history style occurring in large, stable, manmade waterbodies that lack endemic species of herbivorous fish and contain an abundance of plant foods including algae and macrophytes. Similar conditions exist in Tinaroo Falls Dam, and like southeast Queensland impoundments, tilapia in this impoundment appear to grow quickly and reach maturity at a relatively large size.

Altricial life history patterns for *O. mossambicus* are commonly recorded in the literature (Balon 1979, Noakes and Balon 1982, De Silva and Amarasinghe 1989, James and Bruton 1992) and this appears to be the pattern for the Kewarra Beach Drain population and, to a lesser extent, the Herberton Weir population. The Kewarra Beach Drain site is characterised by limited spawning locations, a shallow and harsh environment where water temperature is likely to fluctuate considerably, and where the water level drops dramatically during the drier months of the year. *O. mossambicus* resident at this site have a high mean relative fecundity (three or more times the average relative fecundity at the other sites) and also mature at a smaller size and an earlier age than other populations examined. Similarly in South Africa, James and Bruton (1992) found that small, mature *O. mossambicus* existing in harsh environments were clearly more fecund and had smaller egg sizes than fish living in more benign habitats. While the contrast was not as stark, the Herberton Weir population of *O. mossambicus* also showed some altricial traits, with male and female fish maturing at a relatively small size and age and having a higher relative fecundity when compared with the populations at Tinaroo Falls Dam and Paradise Palms. Additionally, because of their higher altitude, the Herberton Weir sites experience lower winter water temperatures that may have also triggered a tendency towards an altricial life history pattern. James and Bruton (1992) noted that in a South African water body where water levels fluctuated erratically and the temperature regime approached lethal limits during winter, *O. mossambicus* still demonstrated the ability to adopt a precocial life-history style in some habitats. The Lm_{50} values for male and female *O. mossambicus* resident in Tinaroo Falls Dam are relatively high, suggesting a precocial life history pattern. During this study, no overt evidence was found to suggest that *T. mariae* adopted the life history patterns described above for *O. mossambicus* and it does not appear to have been reported in the literature.

The relatively stable and generally low monthly GSIs at the coastal sites (Mulgrave River, Paradise Palms and Kewarra Beach Drain) suggested that spawning occurred throughout most of the year, although slightly higher values in October and November would infer an increase in activity during these months. By contrast, for both *T. mariae* and *O. mossambicus*, reproductive seasonality appeared to be much more pronounced at the cooler, higher altitude sites (Herberton Weir and Tinaroo Falls Dam) than at the warmer coastal sites. The *O. mossambicus* population resident at Herberton Weir and the *T. mariae* population resident at Tinaroo Falls Dam show decreased activity (ie lower GSIs) in the cooler mid-year months increasing to higher levels in the warmer months. The seasonal trend in monthly GSIs for *O. mossambicus* at the Herberton Weir site, albeit with some lag, roughly follows the surface-water temperature patterns. In many parts of the tropics, tilapiine cichlids appear to breed all year round and rising water temperatures seem to be a trigger for spawning (Turner and Robinson 2000). In some areas there is often a peak in spawning activity during the rainy season (Turner and Robinson 2000) and Ikomi and Jessa (2003) related a peak in spawning activity in *T. mariae* to the beginning of the wet season with a minor peak at the height of the floods. In the Townsville area of tropical north Queensland, Webb (1994) found that *O. mossambicus* spawned for 9–10 months and suggested that in the Cairns region, spawning can occur year round. In the more temperate area of southeast Queensland, Arthington and



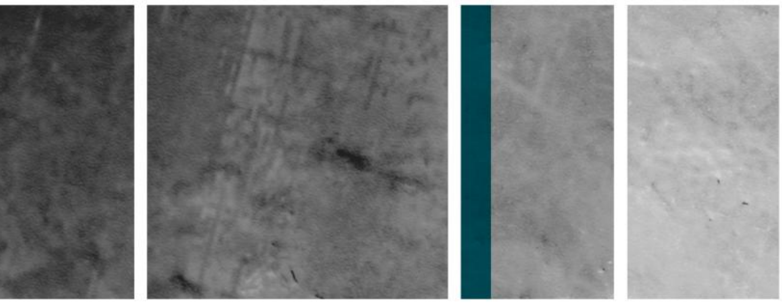
Milton (1986) found that *O. mossambicus* had a seven-month spawning season. Webb (1994) noted that peak spawning activity for *O. mossambicus* occurs just prior to and during the summer wet season when temperatures exceed 32°C.

The magnitude of GSI values, while varying seasonally at individual sites, are also different between sites for the same species. The GSI values for the altricial population of *O. mossambicus* at Kewarra Beach Drain were consistently much higher (up to 4.98 in November 2007) than any of the other populations, suggesting a high level of spawning activity throughout the year. However, this inference is not supported by the occurrence of stage 5 and stage 6 gonads and it may be that, in a markedly stressed population like this, the gonads make up a higher proportion of the total body weight than in other populations. For example, at the nearby Paradise Palms site, the monthly GSI values for the *O. mossambicus* population are all substantially less than those for the corresponding month at the Kewarra Beach Drain site, which peaked slightly in October. Stage 5 and/or stage 6 gonads were present in most months at the Paradise Palms site.

In contrast, both the monthly GSI values and the frequency of stage 5 and/or stage 6 gonads in the samples strongly suggests reproductive seasonality in the *O. mossambicus* populations of Tinaroo Falls Dam and Herberton Weir. In the Herberton Weir population there would appear to be a positive relationship between spawning activity and temperature, with a peak in activity during the warmer summer months. At Tinaroo Falls Dam, with the exception of November and December, the GSIs were low for most of the year and there was a relative absence of stage 5 and stage 6 gonads (except in March), with samples being dominated by stage 2 and stage 3 fish. These data infer either low levels of spawning in the impoundment or that the sampling regime was not robust enough to catch reproductively active fish (ie most fish are spawning elsewhere in the dam or in other parts of the catchment). Only low numbers of juvenile fish were caught at Tinaroo Falls Dam when compared with other locations such as Herberton Weir and Paradise Palms. Most *O. mossambicus* caught in Tinaroo Falls Dam were large fish (n=403, x±SD 343.7 mm ± 69.41 mm), with the smallest individual caught measuring 91 mm. It is unlikely that the apparent low abundance of juvenile fish was related to sampling gear used, as the same gear yielded large numbers of fish < 100 mm in the nearby Herberton Weir sites. Alternatively, the lack of juveniles may be because the Tinaroo Falls Dam population of *O. mossambicus* was tending towards a precocial life-history style where reproduction is delayed and fecundity is reduced (Noakes and Balon 1982).

While the potential and relative fecundity of *O. mossambicus* sampled during this study was highly variable and probably related to life-history style, the fecundity estimates between sites for *T. mariae* were similar. The literature does, however, document some large variations in the absolute fecundities for *T. mariae* from 150–6082 ova (King and Etim 2004). In the current study, the estimates of fecundity (potential fecundity and relative fecundity) for *T. mariae* showed no significant differences between fish sampled at the Tinaroo Falls Dam and Mulgrave River sites. There were, however, significant differences between sites for the four *O. mossambicus* populations. Noteworthy is the relatively high number of eggs/kg observed for fish from the Kewarra Beach Drain population. This is probably further evidence of a plastic phenotypic response to a food-restricted, temperature-stressed and water-quality compromised habitat. At this site, *O. mossambicus* were maturing at an earlier size and the average size of mature female fish was significantly less than the average Lm_{50}'s of mature females from the other sites, suggesting early maturation. Ylikarjula et al (1999) noted that at locations such as this where resources are limited, fish will allocate resources to biomass rather than to growth thereby giving rise to altered life histories including decreased age at maturation. James and Bruton (1992) observed typical altricial life-history style characteristics in *O. mossambicus* at one of their sampling locations, which they attributed to environmental stress. The average relative fecundity for fish sampled at the Herberton Weir sites, while less than that estimated for the Kewarra Beach Drain location, was significantly more than either of the Paradise Palms or Tinaroo Falls Dam populations. With the exception of the Kewarra Beach Drain site, the relative fecundities obtained in the current study were similar to those estimated by James and Bruton (1992) for *O. mossambicus* in South Africa. In southeast Queensland, Arthington and Milton (1986) determined that there were significant differences in the mean non-standardised fecundities of *O. mossambicus* from two different impoundments of 2107 and 1360 respectively. The former is comparable to the mean fecundity of *O. mossambicus* caught in Tinaroo Falls Dam during the current study, while the latter is slightly higher than the PFs estimated for *O. mossambicus* populations at Paradise Palms and Herberton Weir. The advanced reproductive strategy of parental care either in the form of mouthbrooding for *O. mossambicus* or substrate spawning for *T. mariae* (Turner and Robinson 2000) translates to a relatively low fecundity, but also results in higher survival. The number of eggs produced is also related to the size of the female and for populations established in marginal habitats such as the Kewarra Beach Drain, this means that relatively few eggs are produced in very small, but mature fish. As males grow faster and reach larger sizes than females in most tilapiine species (Oliveira and Almada 1995), it is not unexpected that male *T. mariae* and *O. mossambicus* dominated the larger size classes in this current study.

The prodigious, invasive capacity of tilapias has been well documented. In Florida, Annett (1999) observed that *T. mariae* dominated the freshwater fish fauna within 10 years of its introduction. Shafland (1996) also noted that tilapia have become so abundant in Florida canals that they make up more than 20% by total number and total weight of all fishes present. Courtenay and Hensley (1979) were able to directly attribute decreases in the population of some native and exotic species in Florida canals to competition from and population increases of *T. mariae*. Peterson et al (2004) asserted that the reproductive characteristics of *O. niloticus* (including a high level of parental care ensuring that the majority of their eggs will survive to the juvenile stage and the ability to spawn multiple broods throughout a year-round reproductive season) gave them a competitive advantage over native fishes. In this study, feral populations of both *O. mossambicus* and *T. mariae* within about a 67 km radius showed a wide diversity and plasticity in their reproductive parameters that is, in part, indicative of their capacity to quickly and efficiently invade new and sometimes marginal areas. In addition, earlier work on *O. mossambicus* in the same study area suggested that these populations might be endowed with genes from either one or more of *O. niloticus*, *O. aureus* and *O. honorum* (Mather and Arthington 1991), thereby potentially according them hybrid vigour. This possibility, along with the plasticity in life-history traits observed during this study, affords *O. mossambicus* an added advantage over native fishes. This will undoubtedly help ensure the continued existence and progressive range expansion of feral *O. mossambicus* populations throughout northern Australia.



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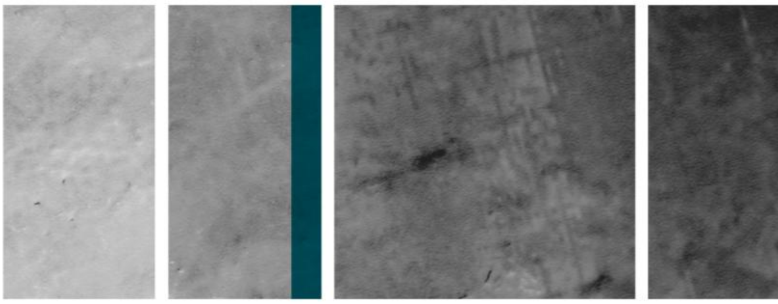

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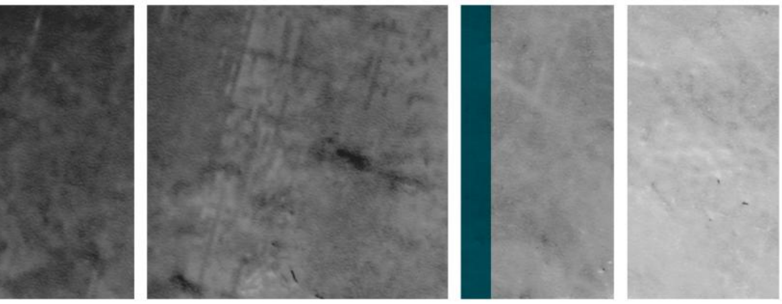


4 Age and growth

4.1 Introduction

Feral populations of two species of tilapia (*Oreochromis mossambicus* and *Tilapia mariae*) are now widely distributed in tropical northeastern Queensland, with *O. mossambicus* also present in southeastern Queensland and some river systems of Western Australia (Arthington 1986, Mather and Arthington 1991, Russell et al 2003, Canonico et al 2005, Webb 2007). It is known that *O. mossambicus* existed in impoundments in southeastern Queensland, as well as urban drains and ornamental ponds in the Townsville region of north Queensland from about the late 1970s (Arthington et al 1984, Arthington and Milton 1986), while the other feral species *T. mariae*, became established in some easterly-flowing tropical streams by the early 1990s (Russell and Hales 1993; Russell et al 1996, 2003). In Australia, feral tilapia are widely regarded as a pest that could have the potential to threaten both native fish stocks and biodiversity (Arthington 1991, Arthington and Bluehdorn 1994, Morgan et al 2004, Canonico et al 2005). In a laboratory study, Doupé et al (2009) speculated that the ecological impacts of *O. mossambicus* might be far more serious than is currently thought and suggested that its reproductive activities might severely disrupt the spawning success of the sympatric native species *Melanotaenia splendida splendida*. Overseas studies have also documented problems associated with feral populations of both *T. mariae* and *O. mossambicus*. For example, in southern Florida, *T. mariae* has become the dominant fish in many water courses where it has been introduced and is seen as the causal factor in the decline of some native and introduced fish species (Courtenay and Hensley 1979). Furthermore, the aggressive behaviour of *O. mossambicus* is also suspected to have caused the collapse of the milkfish (*Chanos chanos*) fishery in Nauru (Fortes 2005), it has invaded Pacific atolls (Lobel 1980) and is controversially thought to have attributed to the near extinction of a native species in Lake Buhi in the Philippines (De Silva et al 2004).

There have been numerous studies investigating the age and growth of *O. mossambicus* across both its natural range in Africa (Bruton and Allanson 1974, Bruton and Boltt 1975, Hetch 1980, Weyl and Hecht 1998) and at locations where it has been introduced, including Okinawa (Tachihara and Obara 2003), Hong Kong (Hodgkiss and Man 1977), the United States (Caskey et al 2007) and Sri Lanka (Amarasinghe et al 1989). Many of these studies have used otoliths to determine the age structure of *O. mossambicus* populations (Hodgkiss and Man 1977, Tachihara and Obara 2003, Caskey et al 2007) while others have used scales (Hodgkiss and Man 1977) and length–frequency analyses (Amarasinghe et al 1989). In contrast, little is known about the age and growth characteristics of *T. mariae,* with most available information gathered from populations surveyed within its natural range in West African coastal drainages in the Gulf of Guinea (Ikomi and Jessa 2003). In Australia, only one analysis of the age and growth of feral tilapia has been undertaken (Arthington and Milton 1986) and this was for *O. mossambicus* populations present in two reservoirs in southeastern Queensland. During this study, Arthington and Milton (1986) did not fit growth curves to their data; however, they did find that *O. mossambicus* grow most quickly during their first year of life. Using scales to age fish, they identified a maximum of four age groups (0+ to 3+ years old) from both reservoirs.

Understanding the population dynamics of the various tilapia stocks within northern Australia is critical to developing effective management strategies for these species. Age and growth data are also necessary variables required for accurate simulation of the effects of management interventions using various computer and statistical models. One such model, CARPSIM (Brown and Walker 2004) has been adapted for use with tilapia (P Brown, Department of Primary Industries, Victoria, personal communication) with its utility reported on elsewhere in this document (see Section 5 — Modelling).

Apart from the early work of Arthington (1986), there is a paucity of empirical population parameter data available for feral tilapia in Australia, which has made implementation of models such as CARPSIM problematic. This current study aims to provide fisheries managers with basic information on local tilapia growth and age structure that can be used directly in planning control programs. It may also be used to parameterise models such as CARPSIM that in turn can be used to gauge the effectiveness of various management and control alternatives.

4.2 Methods

At capture, each fish (both *T. mariae* and *O. mossambicus*) was measured (total length, TL), weighed (in grams) and sexed before the sagittal otoliths were extracted. Only fish more than approximately 80 mm TL had their otoliths removed, with smaller fish assumed to be of age 0+. Otoliths were air dried for up to a week and stored prior to sectioning. Once dry, each sagitta was weighed using an AandD GR200 analytical balance (Milpitas, USA) that was linked to an Excel spreadsheet on a personal computer. Unless broken or lost (in which case the remaining sagitta was used), the right side sagittae were embedded in epoxy resin and cut transversely into approximately 0.5 mm-thick sections using a Gemmaster high speed saw (Shellap Supplies Pty Ltd, Adelaide, Australia). Otolith sections were then mounted in epoxy resin between microscope slides and cover slips and viewed on a black background under reflected light using a stereo microscope (x 10 to x 40 magnification). Images of sections were imported into Optimas (Version 6.5) image analysis software, with measurements taken of the position of each complete opaque band from the nucleus along an axis ventrally adjacent to the sulcus on the proximal edge. As well as these measurements, the reader was recorded and each sectioned otolith given an index of readability on a scale from 1–5 (where 1 was very difficult to read and 5 quite easy). This index was a measure of the confidence the reader had in assigning a count of the number of opaque bands.

The relationship between total fish length and age was fitted to a von Bertalanffy growth formula (VBGF) using the non-linear least-squares method (Excel Solver, Microsoft Corporation, Redmond, USA). This equation is:

$$L_t = L_\infty \left\{ 1 - e^{[-K(t-t_0)]} \right\}$$

L_t is the total length (TL) at age t, and L_∞, K and t_0 are the asymptotic length, Brody growth coefficient and hypothetical age at TL = 0 respectively. Small immature individuals of indeterminate sex were also assigned to the 0+ age group for deriving the pooled VBGF for both sexes.

Marginal increment indices were expressed as a fraction of the last complete increment along the sulcal radius, and were calculated by dividing the total width of the marginal translucent zone by the width between the radii of the two outermost bands. For otoliths with only one annulus, the total width measurement was divided by the radius of the first annulus. Otoliths with an incomplete annulus on the margin were assigned an index of 0. The periodicity at which checks were laid down was also determined by inducing the formation of a permanent mark on the otoliths using oxytetracycline (OTC) and then either releasing the fish back to the wild and recapturing them at a later date, or keeping them in tanks at the Northern Fisheries Centre for periods of up to 438 days. Larger tilapia (> 100 g) were anesthetised in a mild solution of Aqui-S (Aqui-S New Zealand Ltd) at the rate of 5 mg/L before they were injected with OTC at a dosage of 50 mg/kg while smaller fish were bathed in a 500 mg/L OTC solution for a period of 8 hours.

The otoliths of OTC treated fish were thin-sectioned and aged as described above except that they were stored in the dark (OTC is photosensitive) both before and after processing to protect the integrity of the fluorescent band. The presence and position of the fluorescent mark was determined using a Nikon TE300 ultra violet microscope. The annual frequency at which the opaque checks were laid down was determined by their relative position to the florescent band. This was done by comparing the number of checks between the florescent band and the edge of the saggita and the period since the fish had been treated with OTC.

4.3 Results

4.3.1 Otolith readability

At the time of assessment, an index of readability was assigned to otolith sections of 1908 *O. mossambicus* and 1580 *T. mariae* (Figure 12). Overall, only 12% of reads were assigned a high or very high confidence (4 or 5), while more than 30% were given a low or very low confidence (1 or 2). *O. mossambicus* from the coastal locations of Kewarra Beach Drain and Paradise Palms were the most difficult to read, with 33% and 29% of fish respectively having a readability of either 1 or 2. In contrast, of the *O. mossambicus* sampled from the two inland sites of Tinaroo Falls Dam and Herberton Weir only 6% and 17% respectively had readability indices of either 1 or 2. Nearly 54% of *O. mossambicus* sections examined from fish caught in Tinaroo Falls Dam had a high readability of either 4 or 5, while almost 38% of *O. mossambicus* from Herberton Weir also had high readability indices of either 4 or 5.

The readability of *T. mariae* otoliths from fish caught in the Mulgrave River and Tinaroo Falls Dam were 11% and 23% respectively for sections with an index of 1 or 2, and 36% and 31% respectively for otoliths with a high index of readability (4 or 5).

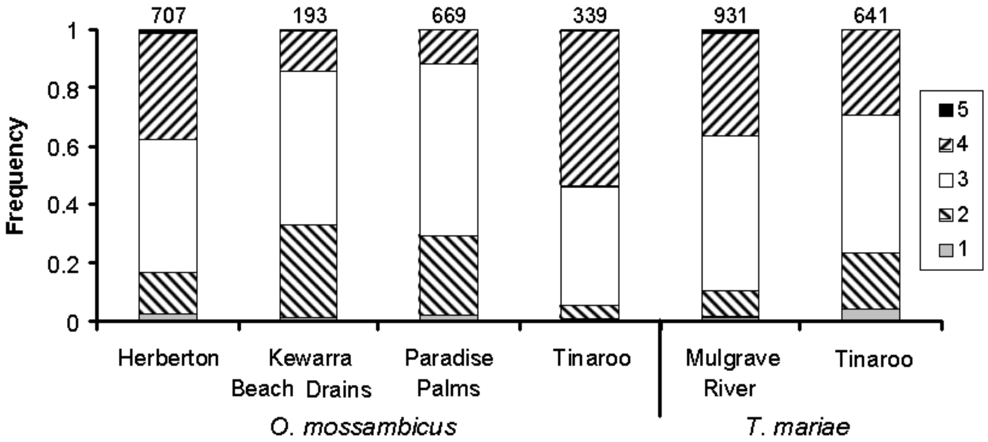


Figure 12: Index of readability for otolith sections from two species of tilapia on a scale from 1–5 (where 1 was very difficult to read and 5 quite easy).

4.3.2 Seasonal pattern of annulus formation

4.3.2.1 Marginal increment analysis

Figure 13 shows the mean monthly marginal-increment indices for all locations. For *T. mariae* in the Mulgrave River and Tinaroo Falls Dam, the increment was narrowest in July and widest in January and May respectively. This pattern was consistent with new annulus formation during the winter months. In the Tinaroo Falls Dam population, there was a sharp drop between May and July, suggesting the formation of a new annulus over a relatively short period (June/July) followed by a period of little or no growth.

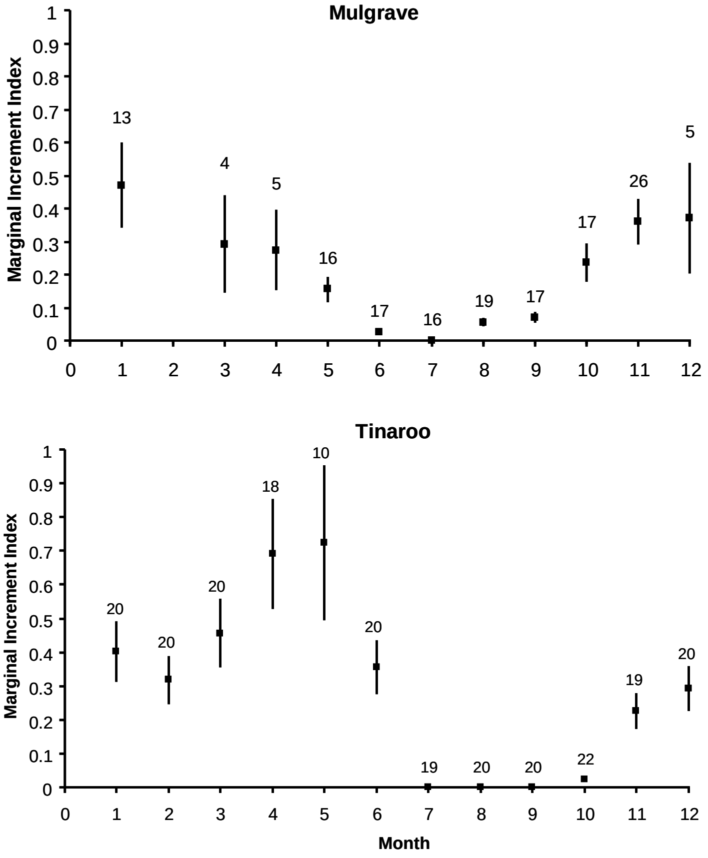


Figure 13: Pooled average monthly increments for *T. mariae* from Tinaroo Falls Dam and the Mulgrave River.

The marginal increment in *O. mossambicus* otoliths from Herberton Weir fish fell sharply at the onset of cooler temperatures from about March and did not begin to increase again until October (Figure 14). There is a similar pattern in *O. mossambicus* from Tinaroo Falls Dam, where there was a sudden fall in the marginal increment indices from June followed by a gradual increase later in the year. For *O. mossambicus* resident at the coastal sites of Kewarra Beach Drain and Paradise Palms, the marginal increment indices fell gradually from about March until about August before beginning to rise.

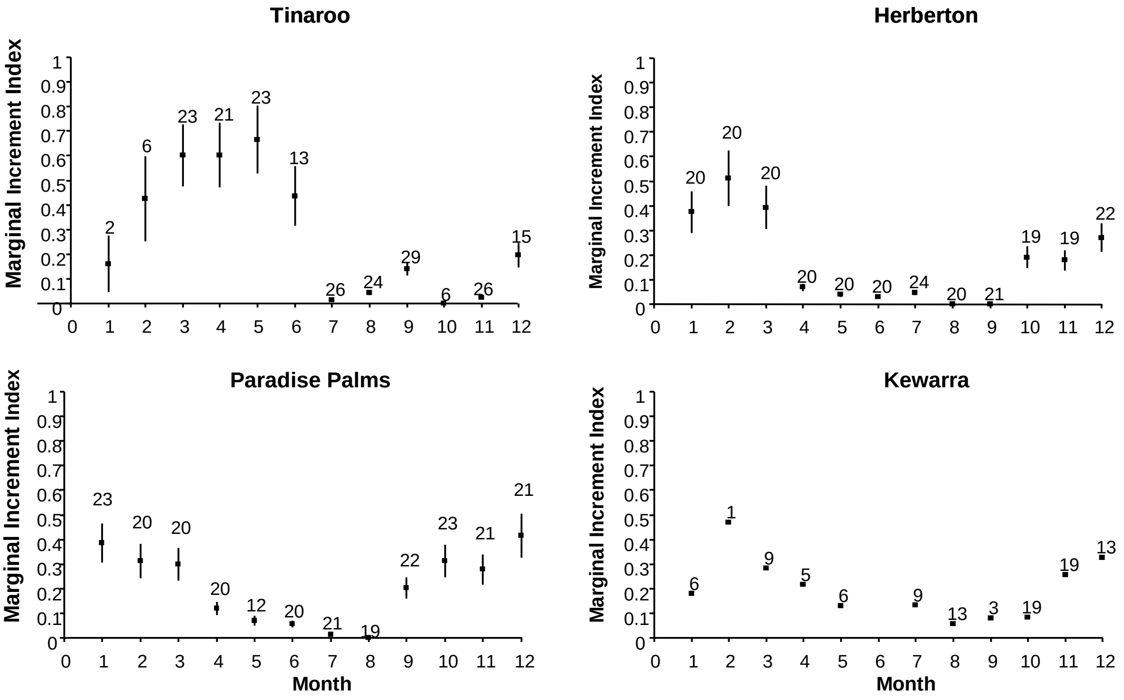


Figure 14: Pooled average monthly increments for *O. mossambicus* from Tinaroo Falls Dam, Paradise Palms, Herberton Weir and Kewarra Beach Drain.

4.3.2.2 Validation of OTC mark

A florescent mark was clearly visible under ultraviolet light in the thin otolith sections from all fish that had been treated with OTC (see Figure 15). The 13 OTC-marked *O. mossambicus* that had been recovered from the coastal lagoon where they were released had been at liberty between 295 and 427 days. Of these, only one fish had no checks between the OTC mark and the margin, 11 had one check and one fish had two checks. The fish that had two checks appeared to have had one recently formed soon after its florescent mark was laid down and another that had just formed before its capture in January 2008. All of the 11 fish with one check had completed laying down that check somewhere between October and January. The single fish with no checks was marked in late November 2006 and had just finished laying down a check before it was injected with OTC. It had been at liberty for the shortest time (295 days).

Eleven OTC-marked *T. mariae* were examined to determine if the checks on their otoliths were deposited annually. Two of these fish were recovered from a farm dam where they had been at liberty for 356 and 357 days respectively. The otoliths of both of these fish had a single check between the OTC mark and the outer edge of the otolith. The remaining nine *T. mariae* were captured from the Mulgrave River in March and April 2006 and marked with OTC before being kept in tanks at the Northern Fisheries Centre for varying periods. Eight of these fish were retained in the tanks between 139 and 223 days before being sampled between 22 August and 18 October 2007. None of these eight fish had a completed check between the OTC band and the otolith margin. The other fish sampled from these tanks were retained for a longer period (264 days) until 28 November 2007 and had recently completed forming a check. In coastal areas, *T. mariae* appear to have completed laying down annual checks towards the end of the year around October to December.

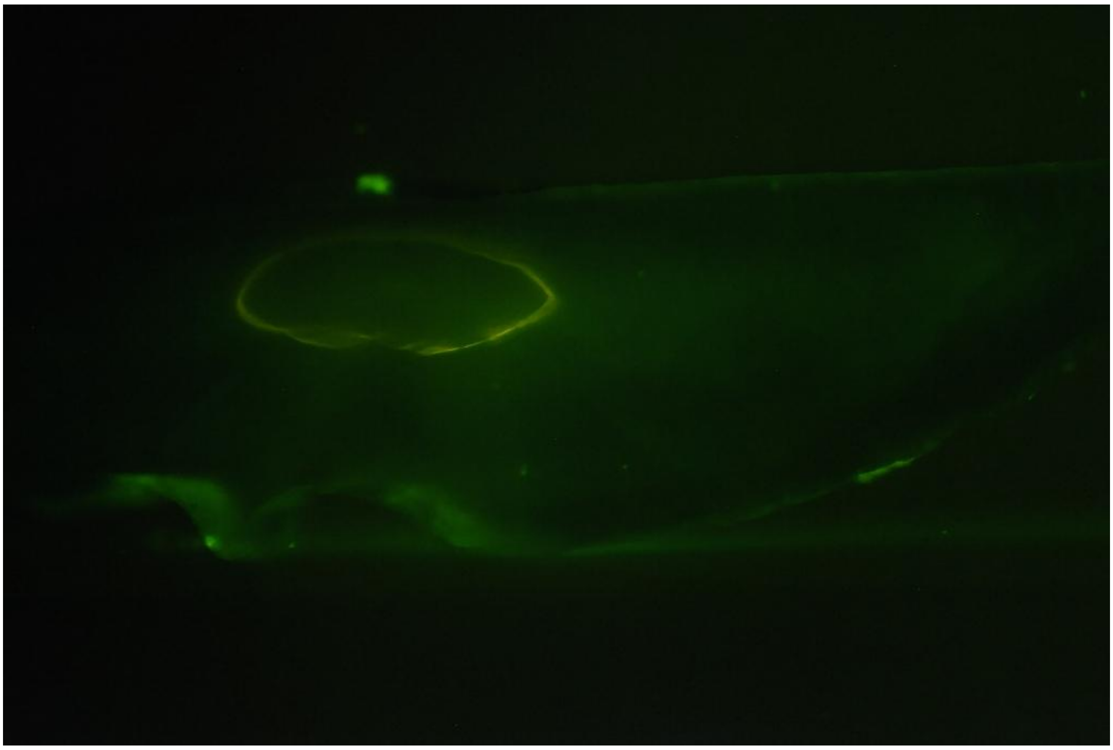
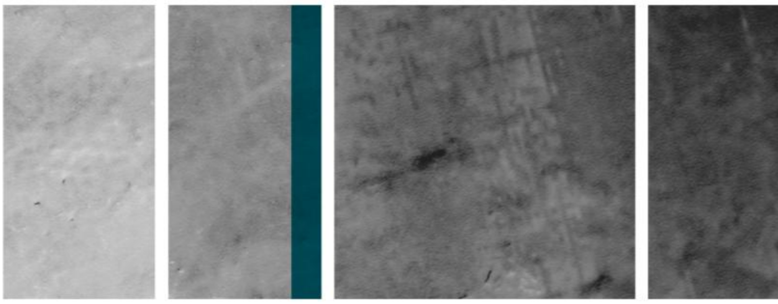


Figure 15: Otolith thin section viewed under ultraviolet light showing a central OTC mark.

4.3.3 Growth

4.3.3.1 O. mossambicus

The fitted von Bertalanffy growth curves and parameters for *O. mossambicus* stocks at the four study sites are given in Figure 16 and Table 4 respectively.



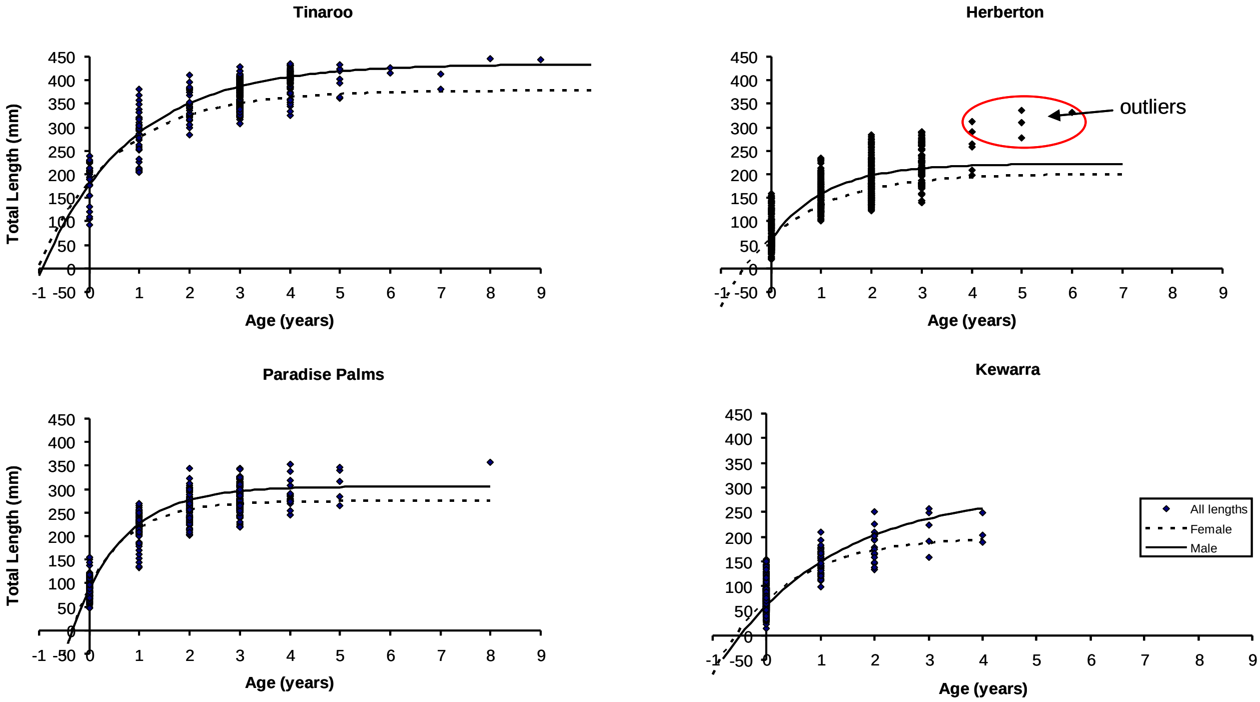


Figure 16: Male and female von Bertalanffy growth curves for *O. mossambicus* from all study sites.

Table 4: von Bertalanffy growth parameters for feral *O. mossambicus* populations from northern Queensland.

	O. mossambicus				
Location	***Sex***	***Number of specimens***	L_∞***(mm)***	***K***	t_0
Herberton	all	934	215.9	0.8	-0.45
	male	640	222.5	0.92	-0.34
	female	618	201.8	0.72	-0.53
Tinaroo	all	320	416.8	0.60	-0.82
	male	194	434.3	0.56	-0.94
	female	147	378.9	0.65	-1.03
Paradise Palms	all	746	288.03	1.08	-0.35
	male	364	295.37	1.16	-0.35
	female	382	273.66	1.09	-0.33
Kewarra	all	937	229.8	0.63	-0.59
	male	708	291.2	0.48	-0.48
	female	803	200.4	0.77	-0.56

Note: ‘na’ is not available

Tinaroo Falls Dam

Assuming that checks are laid annually, *O. mossambicus* up to nine years old were sampled in Tinaroo Falls Dam during this current study. The largest fish was an 8+ year old, 446 mm TL Mozambique tilapia caught in January 2009. For all *O. mossambicus* pooled (males, females and immature fish), the von Bertlanaffy growth parameters (L_∞, K and t_0) were 416.8 mm TL, 0.60 and -0.82 years respectively. Males grew larger than females in Tinaroo Falls Dam (Figure 16). The von Bertalanffy growth parameters (L_∞, K, t_0) were estimated for male *O. mossambicus* (434.3 mm TL, 0.56, -0.94 years) and female *O. mossambicus* (378.9 mm TL, 0.65, -1.03 years) for *O. mossambicus* sampled at this location (Table 4).

Paradise Palms

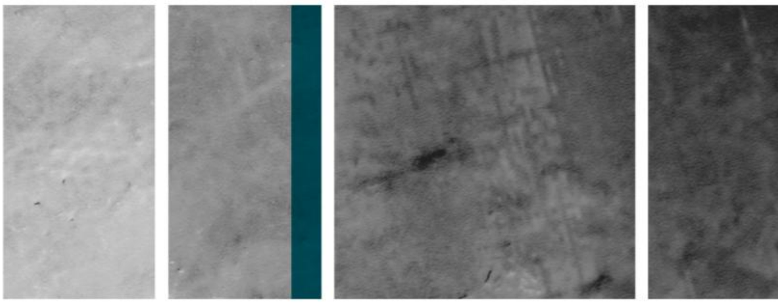
The maximum age of fish sampled in the lagoons at Paradise Palms was 8+ years, although few fish older than 4+ years were captured. The von Bertalanffy growth parameters (L_∞, K, t_0) were estimated for male *O. mossambicus* (305.9 mm TL, 1.01, -0.33 years), female *O. mossambicus* (275.0 mm TL, 1.15, -0.34 years) and all *O. mossambicus* pooled (290.9 mm TL, 1.03, -0.36 years) for *O. mossambicus* sampled at this location (Table 4).

Herberton Weir

The largest fish caught at this site was 335 mm TL with the oldest fish only 6+ years. The length-at-age data for all *O. mossambicus* sampled in the Herberton weir (Figure 16) show some outliers (predominantly female) in the 4+, 5+ and 6+ age groups that appear to be disproportionately large for their age. All larger fish were caught during the first two years of sampling (2006 and 2007). When these data are included in the analyses, the von Bertalanffy growth parameters (L_∞, K, t_0) for males are 222.48 mm TL, 0.92, -0.34 years, for females are 375.33 mm TL, 0.23, -0.83 years and for all *O. mossambicus* pooled are 256.8 mm TL, 0.53 and -0.57 years. The L_∞ value for females appeared very large and may have been disproportionately influenced by the abovementioned outliers. When these outliers were removed from the analyses, the von Bertalanffy growth parameters (L_∞, K, t_0) for female fish diminished substantially to 201.8 mm TL, 0.72, -0.53 years), but the parameters for the male fish remained the same (Table 4). The von Bertalanffy growth parameter estimates (L_∞, K, t_0) for all samples pooled (excluding the outliers) were 215.9 mm TL, 0.80 and -0.45 years respectively.

Kewarra

The oldest fish sampled at Kewarra was 4+ years and most of the fish sampled at this location were from three age classes (0+ to 2+). The largest fish was only 256 mm TL. There was a difference between the von Bertalanffy growth parameters (L_∞, K, t_0) for male (291.2 mm TL, 0.48, -0.48 years) and female (200.4 mm TL, 0.77, -0.34 years) *O. mossambicus* sampled Table 4). For all samples pooled regardless of sex, the von Bertlanaffy growth parameters (L_∞, K, t_0) were 229.8 mm TL, 0.62 and -0.36 years respectively.



4.3.3.2 *T. mariae*

The fitted von Bertalanffy growth curves and the population age structure for *T. mariae* resident in the Mulgrave River and Tinaroo Falls Dam were different in a number of ways (Figure 17, Table 5). For example, in the Mulgrave River the maximum age of fish sampled was 8+ years while in Tinaroo Falls Dam the maximum age was only 4+ years. In Tinaroo Falls Dam, the L_∞ for all fish pooled was 252.9 mm TL with a relatively high *K* value of 1.36. As a result, early growth was rapid, with 1+ year-old fish approaching asymptotic length. *T. mariae* in the Mulgrave River had a lower *K* value (0.57) and consequently took longer to approach a higher L_∞ of 278.7 mm TL. In both study sites, male *T. mariae* grew larger than females.

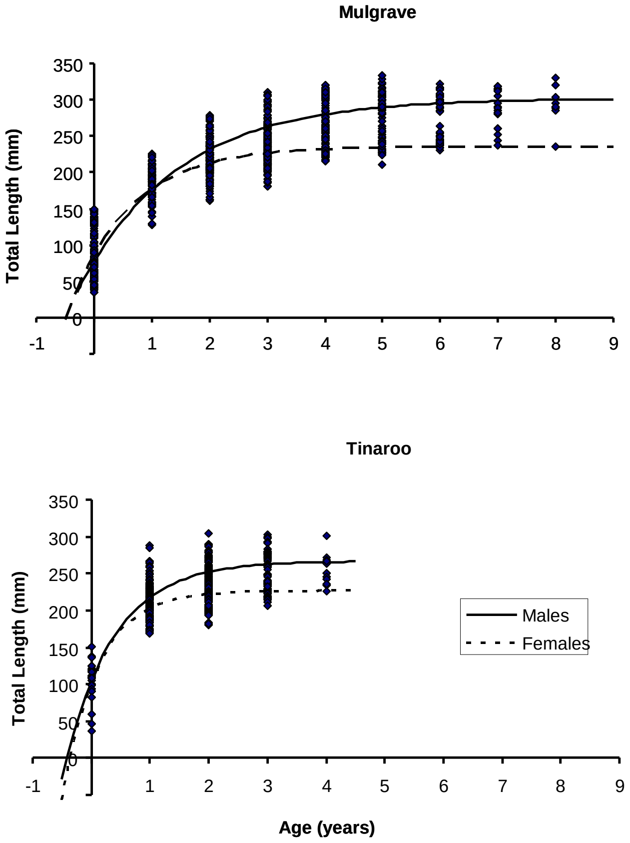


Figure 17: Male and female von Bertalanffy growth curves for *T. mariae* from all study sites.

Table 5: von Bertalanffy growth parameters for feral *T. mariae* populations from northern Queensland (immature fish of indeterminate sex were included in data used to calculate values for males and females).

	T. mariae				
Location	***Sex***	***Number of specimens***	***L_∞(mm)***	***K***	***t_0***
Mulgrave River	all	945	287.7	0.57	-0.7
	male	524	302.1	0.58	-0.51
	female	574	236.0	0.93	-0.48
Tinaroo	all	481	252.9	1.36	-1.08
	male	266	274.6	0.81	-0.91
	female	241	230.5	1.02	-0.99

Note: 'na' is not available

4.4 Discussion

A number of factors are thought to influence opaque zone formation in otoliths by slowing the somatic metabolic rate of the fish. In tilapia, these factors include seasonal changes in temperature, temporal variation in food quality and courtship activities (Weyl and Hecht 1998, Bwanika et al 2007). Many tropical fish species, including cichlids, are notoriously difficult to age using hard bony structures (Abowei and Davies 2009) because of the annual uniformity of their environmental conditions. In this current study, the readability of otoliths of the same species taken from four geographically close sites was found to be variable. *O. mossambicus* sampled in coastal areas had a higher proportion of difficult-to-read sections (with a readability index of 1 or 2) than those fish taken from the higher altitude sites (ie Tinaroo Falls Dam and Herberton Weir — where there is a bigger differential between winter and summer water temperatures). The 'easier' readability of *O. mossambicus* otoliths from Tinaroo Falls Dam infers that water temperatures (or some related factor) might play a role in determining when checks are formed. Marginal increment analyses suggested that the growth check is laid down sometime after mid-year in this species. In coastal areas, both the marginal increment analyses and the age validation experiments using OTC permanent marks suggested that *O. mossambicus* and *T. mariae* lay down only one check per year, probably during the spring and summer from about October to as late as January.

In contrast with fish from upland areas (Tinaroo Falls Dam and Herberton Weir), the otoliths of *O. mossambicus* resident in coastal areas such as Paradise Palms and Kewarra Beach Drain were more difficult to read. This was probably because these fish were from a homogenous coastal environment with little annual temperature variability (Abowei and Davies 2009) and where primary productivity (and therefore food supply) was enhanced through the addition of nutrients from the surrounding urban areas (Kewarra Beach), or effluent runoff from the adjacent golf course (Paradise Palms). In the relatively benign environmental conditions of Tinaroo Falls Dam and Paradise Palms, *O. mossambicus* populations were well established with fish up to 8+ years sampled in both locations during this current study. However, in the more variable environmental conditions at Kewarra Beach, *O. mossambicus* were 'stunted' and the oldest fish caught was 3+ years, with most fish between only 0+ and 2+ years.

‘Stunting’ is a well-established trait in *O. mossambicus* populations and is thought to be primarily a phenotypic response (Lorenzen 2000). In support of this argument, Lorenzen cites as evidence the rapid emergence of ‘stunted’ phenotypes following the isolation of sub-populations from non-’stunted’ stocks. He also points out that in other species, ‘stunting’ has been shown to be reversible through experimental manipulation. Growth characteristics of *O. mossambicus* stocks sampled during this study were highly plastic with the variation observed in L_∞ suggesting a continuum between ‘normal’ larger growing populations such as those in the expansive waters of Tinaroo Falls Dam, and smaller crowded populations such as those observed in the harsher environmental conditions of the Kewarra Beach Drain. Lorenzen (2000) noted that the environmental conditions most regularly associated with ‘stunting’ in tilapias included where there was crowding in small water bodies, where the populations were isolated and when the local fish community was at a low diversity. Certainly all these conditions apply to some extent to the Kewarra Beach Drain stocks and, to a lesser degree, the Herberton Weir populations where ‘stunting’ is most apparent. ‘Stunting’ is discussed in further detail in the Reproductive strategies section.

The growth curves for *T. mariae* in Tinaroo Falls Dam and the Mulgrave River were very different, with the latter location having a higher L_∞ (by ~10%) and lower *K* value. In Tinaroo Falls Dam, only five age classes (0+ to 4+) of *T. mariae* were recorded compared with nine age classes (0+ to 8+) in the Mulgrave River. This was probably because the Tinaroo Falls Dam population was much newer and thus less well established than the Mulgrave River population, which has been known to exist from at least the early to mid-1990s (Russell et al 1996). By contrast, the Tinaroo Falls Dam population had only been known since the early 2000s (Alf Hogan, DEEDI, personal communication). Surveys in the 1990s found that there were well-established populations of *O. mossambicus*, and to a lesser extent *T. mariae*, present in farm dams and tributaries of the upper Barron River catchment, with fish from these areas probably dispersing downstream to colonise Tinaroo Falls Dam (Webb et al 1997, Russell et al 2003). Taking into account that these surveys suggested that *O. mossambicus* was the most abundant species at that time, it is feasible that *T. mariae* would colonise Tinaroo Falls Dam later than *O. mossambicus*. Given the high value of Brody’s growth coefficient (*K*) for the Tinaroo Falls Dam stock of *T. mariae*, it is not surprising that their growth curve approaches asymptotic length within two years. Such rapid early growth may be extremely important when colonising a new environment, as it allows juveniles to quickly attain a size large enough to minimise predation and simultaneously reach sexual maturity to enable reproduction (Weyl and Hecht 1998). In a Nigerian rainforest stream, King and Etim (2004) calculated an L_∞ for *T. mariae* of 304 mm and a growth coefficient *K* of 0.4/year. It may therefore be that as the *T. mariae* population in Tinaroo Falls Dam becomes more established and further data for older age classes becomes available, estimates of the growth parameters of its tilapia population may converge on those estimated for fish in the Mulgrave River. Indeed, as both populations become further established, their von Bertanffy growth parameters might more closely resemble values for fish in their natural range.

In many tilapiine species males grow faster than females (Faunce et al 2002). Faunce et al (2002) speculated that sperm production required less energy than egg production and that the slower growth observed in females was due to differences in energy budgets during the spawning season. Jiménez-Badillo (2006) noted that energy expense is very intense during spawning and this has an important effect on growth. This is exacerbated in tropical areas such as north Queensland where many species, including tilapia, have extended spawning seasons. Jiménez-Badillo (2006) further speculated that larger males could have a selective advantage over smaller males by being able to defend the brood or a spawning pit more effectively against predation.

The large, 4+ to 6+ year-old female fish caught in the first two years of sampling in the Herberton Weir appeared to have a disproportionate influence on the shape of the female von Bertalanffy growth curve for this population. The available evidence suggests that colonisation of the Herberton Weir by *O. mossambicus* probably occurred within about the last 10 years and that these ‘outliers’ may have been either part of the original batch of fish stocked into the weir or their immediate progeny. Furthermore, there is evidence of ‘stunting’ in subsequent generations that would have affected the overall growth parameters of this population (see Reproductive Strategies section).

Even though all sites where *T. mariae* and particularly *O. mossambicus* were sampled during this current study were geographically close, the growth characteristics of the various populations were highly variable. This plasticity in growth, as well as other life-history metrics (eg reproductive characteristics including parental care and age and size at maturity) enhances the ability of tilapia (and cichlids in general) to invade and become established in non-native environments (Peterson et al 2004). It also highlights the need to carefully consider which population parameters should be used when modelling the impacts of control measures on feral stocks of tilapia, particularly *O. mossambicus*.

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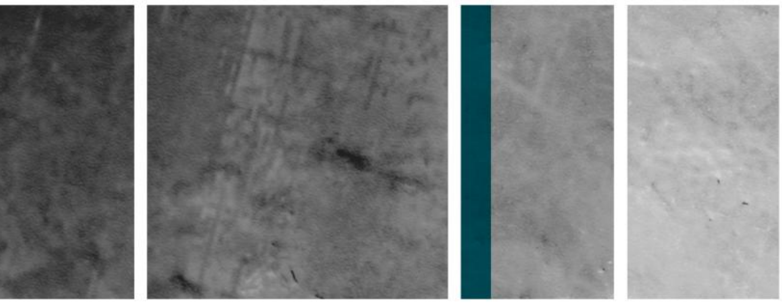


5 Modelling

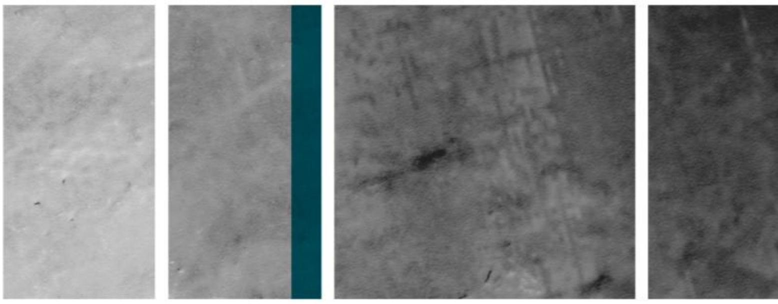
5.1 Introduction

The capability to simulate the effects of various management strategies provides not only an effective tool for managers to plan and implement control strategies, but is also an excellent education and extension mechanism. It allows the likely impacts of management interventions or changes to major environmental variables to be graphically demonstrated to a wider audience of interested parties, including community and natural resource management groups. This was suggested by Murie et al (2009) who recognised the potential benefit that models may have in quantifying the effects of major environmental and biotic variables on populations of an apex fish predator in the Florida Everglades. While pest control models are widely known (eg Sharov 1996), there are also models available that simulate population dynamics to evaluate options for sustainable resource management, including fisheries management (Beamesderfer 1991, Sharov 1996, Xie et al 1999, Brown and Walker 2004). Brown and Walker (2004) developed CARPSIM, a simple age-based model, to simulate the effects of a range of management scenarios on feral carp populations in southern Australia. Carp management scenarios that they successfully simulated included the effects of fishing on spawning stock, fishing the whole stock, spawning or recruitment sabotage, and driving the population sex ratio towards male dominance. Given the success and utility of the CARPSIM model, it was decided to modify its code for use with feral populations of *T. mariae* and *O. mossambicus,* using population parameters derived during this study for resident populations in northern Queensland.

5.2 Methods

CARPSIM software was modified by collaborator and model developer, Paul Brown. These modifications included adding features that improved its utility and its capacity to be used with other fish species, including tilapia. The first version that was used to develop tilapia models was CARPSIM 2.0.2, although CARPSIM 2.0.8 is the version currently being used. To facilitate the development of the tilapia models, project staff met three times during 2008.

The first priority was to develop working models for single populations of *T. mariae* and *O. mossambicus*. These models were developed using population parameters obtained from work conducted with various stocks earlier in the current study. Details of how these parameters were obtained are given in the Reproductive Strategies and Age and Growth sections of this report. Once modelling commenced, it quickly became apparent that some of the data being collected as part of the routine sampling program were inadequate to develop workable stock-recruitment relationships for either of these species. Recruitment in the CARPSIM model was assumed to be density dependent, whereby high densities of spawning adults can reduce recruitment through habitat destruction, and competitive inter- and intra-year class interactions (Brown and Walker 2004). Recruitment in the model was derived from a Ricker stock-recruitment relationship between density of yearlings and density of mature adults. The data that were initially collected resulted in a substantial underestimation of the density of recruits at each location. As a result, in late 2007 the sampling program was modified so that at each sampling event a more accurate estimate of the density of recruits was obtained. Workable stock-recruitment relationships have now been established for the Herberton Weir population of *O. mossambicus* and the Mulgrave River population of



T. mariae. It is for these two populations that the effects of various management interventions have been modelled.

5.3 Results and discussion

5.3.1 Simulated management scenarios

The model involved starting with a base population of 50 fish and then, over a series of years, the population of fish would increase naturally until it reached equilibrium. This pattern can be seen for the *O. mossambicus* population simulated, with no interventions, in Figure 18. The output of the model for *T. mariae* is slightly different and shows densities increasing rapidly and then oscillating slightly for a number of years before reaching equilibrium (Figure 18). As the model is further refined, these oscillations may be dampened or disappear completely.

In its present form, the CARPSIM model allows for the simulation of a number of management interventions for tilapia, including fishing, poisoning, recruitment sabotage and predation (on juveniles and adults). The simultaneous use of some of these management interventions can also be modelled. Some examples showing the predicted consequences of various management interventions follow.

5.3.1.1 Fishing pressure using trapping and netting

By setting a fishing mortality parameter (*F*) in the model, the effects of various levels of fishing pressure can be simulated. Removal-based strategies that this modelling simulates include netting and trapping. In the absence of fishing mortality, the density of a newly established population of *T. mariae* (initial population size n=50) quickly increased and stabilised at a density of about 3500 fish per hectare (Figure 18). Similarly, a newly established population of *O. mossambicus* (based on population parameters from the Herberton Weir stock) rapidly increased to over 800 fish per hectare before stabilisation.

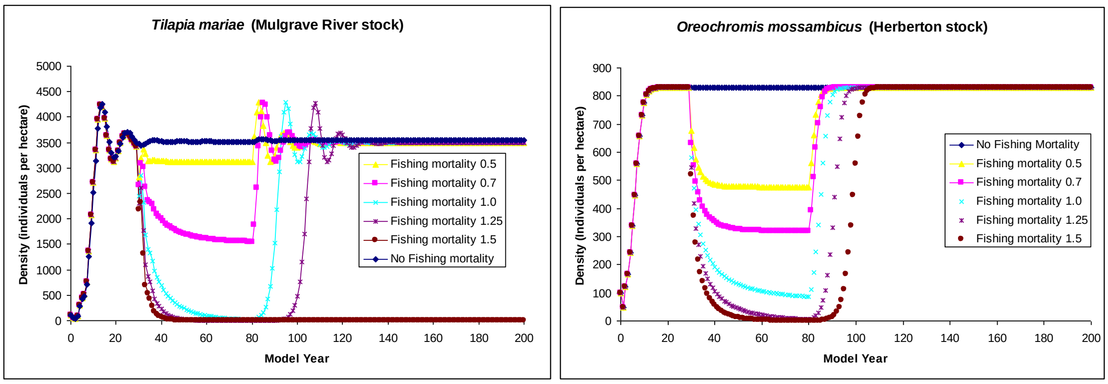


Figure 18: Predicted effects of various levels of fishing mortality on the densities of stocks of *T. mariae* and *O. mossambicus*.

Moderate levels of fishing effort (eg *F*=0.5, 0.7) applied to both species between years 30 and 80 resulted in a temporary fall in fish density. The populations of both species recovered quickly to equilibrium levels once fishing effort was removed from the model. Very heavy fishing effort (eg *F*=1.0, 1.25) applied between years 30 and 80 to both species populations, resulted in a substantial drop in fish density (approaching pseudo-extinction levels). Again, the populations of both species eventually recovered, albeit over a longer timeframe than those subjected to lower fishing mortalities (Figure 18). It was only when a very high fishing mortality of *F*=1.5 was applied to the *T. mariae* population that it was driven to a level where it became pseudo-extinct. Pseudo-extinction is achieved (ie. zero recruitment) in the model when fish density reaches ≤ 0.001 fish per hectare. The *O. mossambicus* population did not become extinct even at this high level of fishing mortality and had completely recovered by about year 100. In practice, such a heavy, sustained fishing effort is probably not achievable and certainly not over a 50-year period.

5.3.1.2 Fishing pressure using angling

Figure 19 shows a simulation of effects of angling on tilapia densities. Two levels of angling effort (*F*=0.5 and *F*=1.0) were modelled. In this example, gear selectivity was decreased by dropping the lengths at which 50% and 90% of the population were caught to 50 mm and 70 mm respectively (ie most of the stock was available for fishing). In the model, the level of angling effort was sustained over 50 years from year 30 to year 80. Applying an angling effort of *F*=0.5 over the 50-year period resulted in tilapia density dropping by about half in both species, but after the cessation of angling, the populations recovered to base levels in about 5–10 years. An angling effort of *F*=1.0 (very high), while resulting in a very substantial decline in tilapia density, did not cause pseudo-extinction of populations. Populations that were subjected to higher fishing mortalities (Figure 18) took longer to recover than those affected by angling mortality.

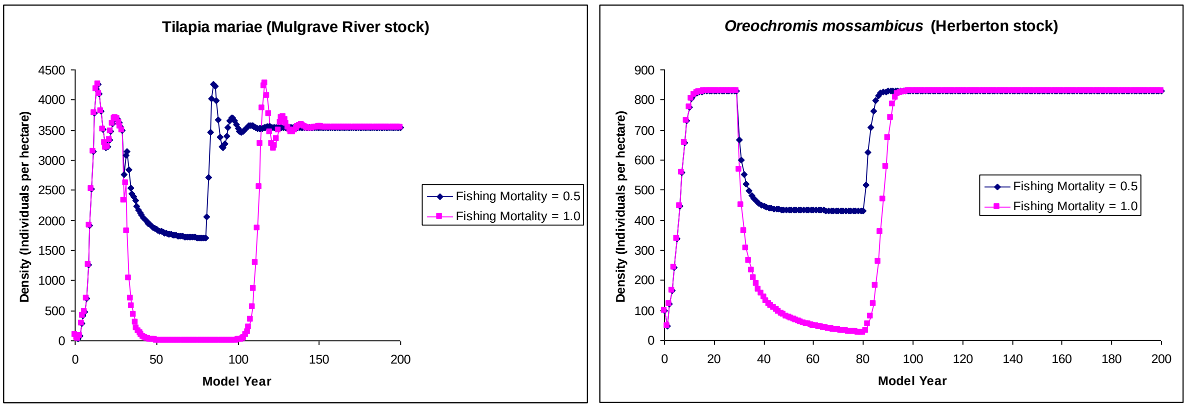
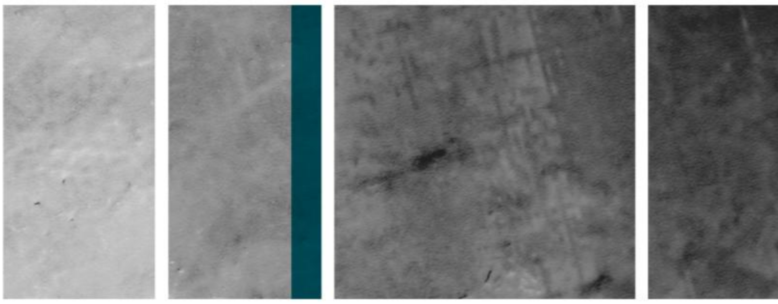


Figure 19: Predicted effects of different levels of angling on the densities of stocks of *T. mariae* and *O. mossambicus*.

5.3.1.3 Single event, large magnitude population reductions

CARPSIM can be used to model the effectiveness of single, even population reductions such as the application of piscicides as a control measure for fish populations. This is done by intervening in a single year (year 30 in Figure 20) through the use of a very high level of the fishing mortality (eg *F*=5.0) and increasing the gear selectivity by dropping the length at which 50% and 90% of the population are selected.



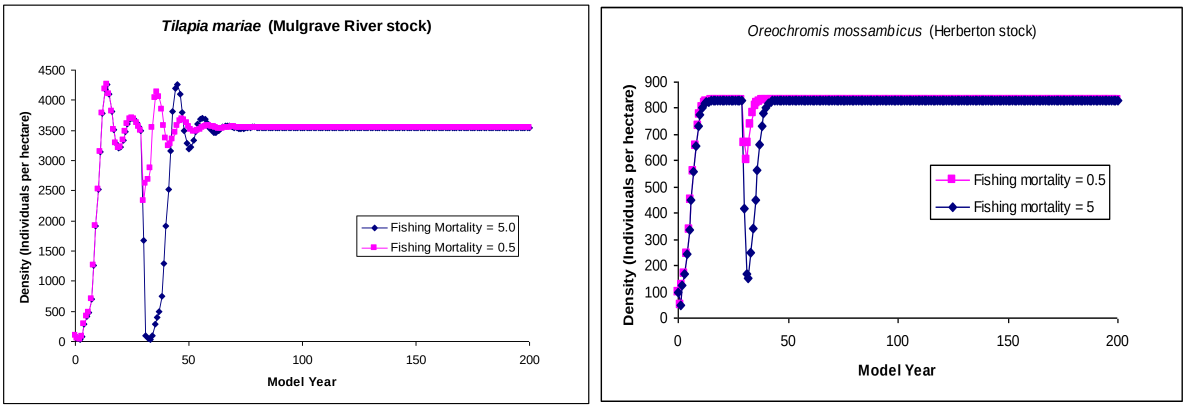


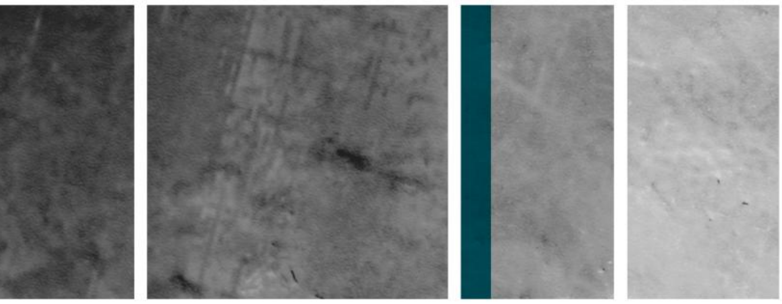
Figure 20: Predicted effects of single event, large magnitude population reductions on the densities of stocks of *T. mariae* and *O. mossambicus*.

Figure 20 shows both the effects of applying a very high level of fishing effort (*F*=5.0) and a lesser level of fishing effort (*F*=1.0) in a single year to both *T. mariae* and *O. mossambicus* populations. These levels of fishing effort were not sufficient to drive populations of either species to pseudo-extinction. While the application of high fishing effort in the model does result in a substantial reduction in fish density (particularly for the *T. mariae* population), over approximately 15 years fish density returns to an equilibrium level of around 3500 *T. mariae* and around 800 *O. mossambicus* per hectare. The recovery assumes that no immigration occurs from surrounding areas. This result suggests that if eradication of either species through the application of a piscicide was to be successful, a single application might not be sufficient so multiple doses could be needed. Brown and Walker (2004) noted that when modelling carp populations with CARPSIM, unselective removal, such as poisoning or trapping all age classes, were more likely to cause pseudo-extinction at levels of *F* >0.7. This suggests that carp populations may be more vulnerable to application of these techniques than are the tilapia populations modelled in the current study.

5.3.1.4 Spawning or recruitment sabotage

The CARPSIM model also allows for the effects of various spawning and recruitment sabotage strategies to be simulated. Brown and Walker (2004) suggested that for carp, these strategies might involve barring access to spawning substrate or hydrological manipulation of water levels to cause stranding of eggs. Similarly for tilapia, a strategy may be to drop the water level in an impoundment to expose spawning pits or to manipulate water levels to minimise the cover provided to juveniles by littoral vegetation. Such strategies might be thwarted by stochastic environmental events such as floods, so that each year there is a risk that spawning sabotage strategies will not be achieved. This possibility is randomly factored into the CARPSIM model.

Figure 21 illustrates scenarios where the probability that recruitment is totally prevented is randomly drawn from a uniform distribution of 25, 50 and 75% of years (R_{fail}=25, 50 or 75%). The model suggests that at all levels of spawning sabotage simulated tilapia populations persist, albeit at depressed or very low density levels. Near pseudo-extinction is achieved for *O. mossambicus* with an R_{fail} of 75%; however, sufficient numbers of fish would still remain to allow the population to recover over time (about 30 years).



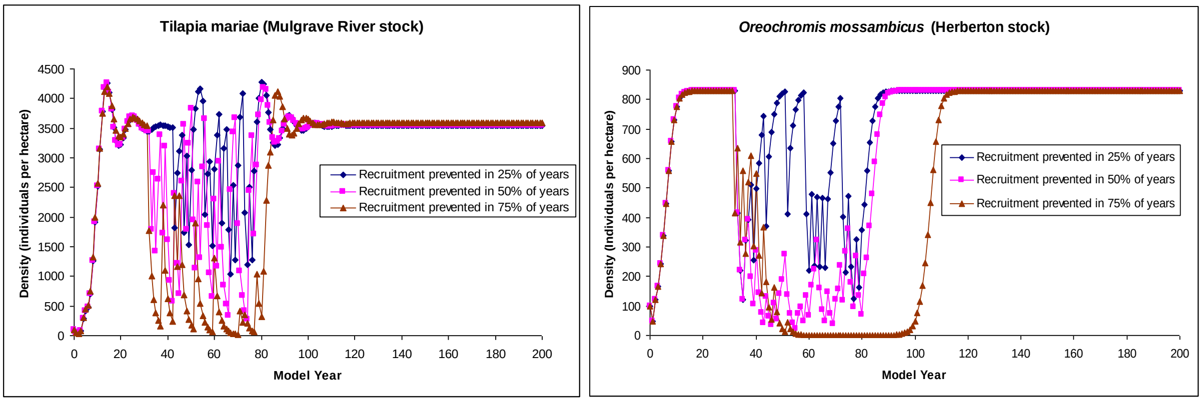


Figure 21: Predicted effects of different levels of recruitment failure on the densities of stocks of *T. mariae* and *O. mossambicus*.

For carp, Brown and Walker (2004) also found that populations persisted where the R_{fail} value was <80%. If recruitment variability was added to the model, the chance of achieving pseudo-extinction via spawning sabotage increased further. CARPSIM uses an assumed relationship between recruitment success and wetter-than-average climatic conditions (ie using the southern oscillation index as a surrogate weighting factor for recruitment) and, as discussed above, this may also be applicable to tilapia. The addition of recruitment variability into a model for an *O. mossambicus* population subjected to an R_{fail} of 75%, resulted in an 80% chance of reaching pseudo-extinction within 46 years of starting the spawning sabotage strategy (Figure 22). For *T. mariae*, when using an R_{fail} of 95%, there is an 80% chance of achieving pseudo-extinction within 44 years of commencing a spawning sabotage strategy (Figure 22).

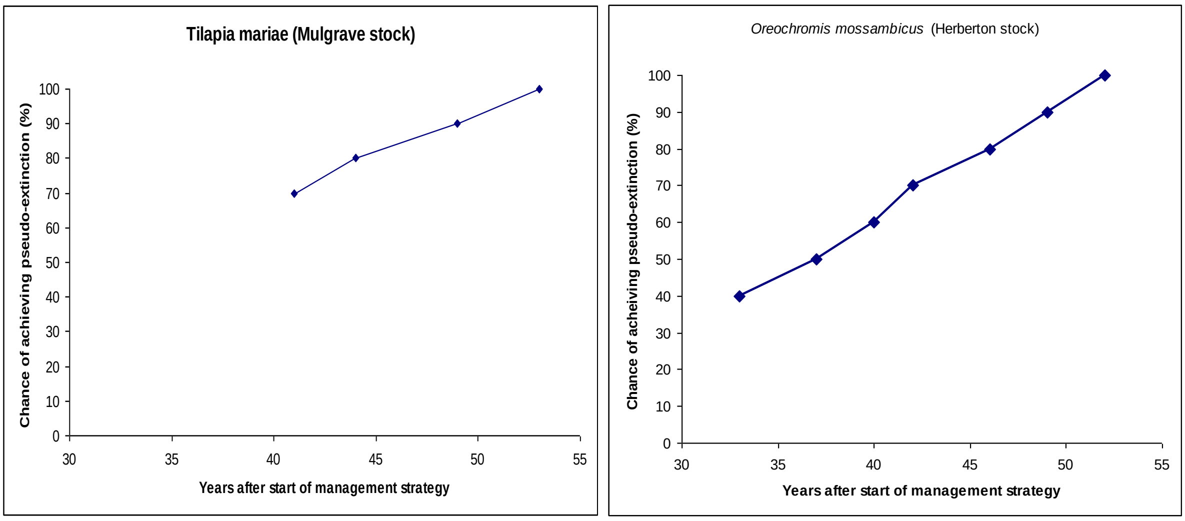


Figure 22: Predicted chance of success for spawning-sabotage strategies (R_{fail}=0.75 for *O. mossambicus* and R_{fail}=0.95 for *T. mariae*) when stochastic trials include recruitment variability (n=100 trials).

5.3.1.5 Integrated management — Spawning sabotage and fishing effort

The above simulations suggest that either very high levels of fishing mortality or very high levels of recruitment failure are required to achieve pseudo-extinction in both *T. mariae* and *O. mossambicus* populations. It may, however, not be practical (or possible) to sustain such high levels of either fishing effort or spawning sabotage individually to achieve this result. But it could be possible to implement a sustained integrated management strategy using lower levels of both fishing effort and recruitment failure over an extended period. Figure 23 shows a simulation of an intervention that included both fishing effort (*F* held constant at 0.7) and different levels of recruitment failure (R_{fail}=10, 25 and 50%). The addition of moderate levels of recruitment failure to the model decreased tilapia density commensurate with the R_{fail} value. While *T. mariae* density was driven down to low levels when interventions were in place, pseudo-extinction was not achieved with any of the parameter levels trialled. However, modelling of *O. mossambicus* populations did achieve pseudo-extinction when recruitment failure was set at 50% (R_{fail}=50%). Where pseudo-extinction was not achieved, the time taken for both the *O. mossambicus* and *T. mariae* populations to return to an equilibrium level was commensurate with the severity of the recruitment failure. The addition of recruitment variability (n= 100 trials) allowed less stringent management interventions to achieve success, with a 90% chance of an R_{fail} of only 25% combined with a fishing mortality level of *F*=0.7, achieving pseudo-extinction within 29 years.

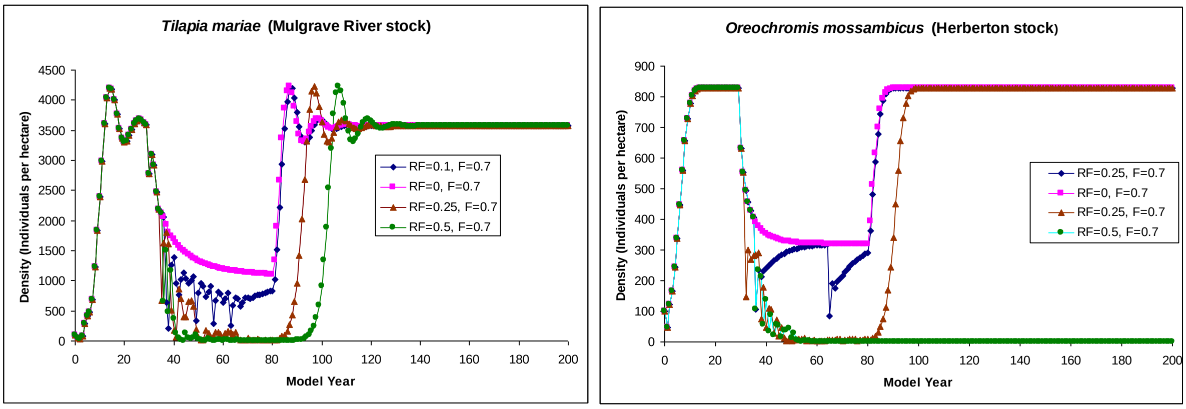


Figure 23: Predicted effects of using two types of interventions, fishing and different levels of recruitment failure, on the densities of stocks of *T. mariae* and *O. mossambicus*.

5.4 Conclusions

The CARPSIM model was successfully adapted for use on populations of two species of feral tilapia that occur in northern Australia. The simulation examples shown above suggest that it is possible to drive *O. mossambicus* and *T. mariae* populations to pseudo-extinction using very high levels of fishing pressure and/or a combination of fishing pressure and recruitment failure. These examples are, however, extraordinary and the simulation modelling undertaken generally suggested that although populations can be driven down to low levels, they recover relatively quickly after management interventions cease. Brown and Walker (2004) also noted that when using CARPSIM on carp populations, the probability of any removal-based method achieving <10% of virgin biomass was small for *F* <1.4. This result suggests that removal-type control measures might be more suited to slower-growing long-lived species whereas faster-growing short-lived fish (such as tilapia) might respond better to spawning sabotage or male-dominance approaches. While the latter approach is currently beyond the scope of this study, it is possible that if the genetic manipulation technology currently being developed for use with carp is successful (Thresher and Bax 2003, Thresher et al 2009), it could be potentially very useful for control of feral tilapia populations. While CARPSIM does allow the modelling of sex ratios biased towards male dominance in carp, this was not achieved for tilapia in this current study. It would be useful to further modify the software to enable CARPSIM to simulate male-dominance scenarios within tilapia populations.

Models such as CARPSIM are extraordinarily useful as education and extension tools. Simulation models showing the effects of various management interventions have already been demonstrated to state government fisheries managers, and the feedback from this exercise was very positive. It is planned to build on this initial showing through a series of in-house seminars to further demonstrate the utility of this model to fisheries managers, research staff and interested community groups.

5.5 Further development

The CARPSIM model is currently still under development by DPI Victoria and future versions would benefit by being able to simulate other types of interventions (eg the addition of predators, both juvenile and adult). The use of a wider range of integrated control measures and inclusion of male dominance strategies to reduce population numbers would also greatly enhance the utility of the program.

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6 Mozambique tilapia control — Herberton Weir case study

6.1 Introduction

Feral Mozambique tilapia populations were first reported from north Queensland during the 1970s and have since been progressively spreading into new catchments throughout the region (Arthington et al 1984; McKay 1984; Arthington 1986, 1988, 1989; Lear 1989; McKay 1989; Russell and Hales 1993; Russell et al 1996, 2000, 2003). The first report of Mozambique tilapia from the top Herberton Weir on the Wild River was in November 2003, and was confirmed later in the same month (Hogan and Vallance 2004). Hogan and Vallance sampled six fish, some of which were brooding females, observed several dozen others, and noted the presence of circular, nest-like breeding leks on the substrate. Hogan and Vallance (2004) also sampled the adjacent bottom weir but found no evidence of Mozambique tilapia either there or further downstream. Between November 2003 and March 2004, Hogan and Vallance removed over 1000 *O. mossambicus* and introduced 120 *Leiopotherapon unicolor* (spangled perch) as a predator control measure, targeted at reducing the number of juvenile Mozambique tilapia present in the weir. *L. unicolor* was present naturally in the bottom weir and in other parts of the Wild River, but not in the top weir. In a later survey, Peverell et al (2005) noted that a population of *L. unicolor* appeared to be established in the top weir but they could find no evidence that they were preying on juvenile Mozambique tilapia. Hogan and Vallance (2004) also noted the presence of *Hephaestus fuliginosus* (sooty grunter) in the top weir, which they concluded was illegally stocked, perhaps at around the same time as *O. mossambicus*.

Over time there was evidence that *O. mossambicus* was spreading downstream, firstly into the bottom weir and then into the Wild River. *O. mossambicus* was first reported in the bottom weir by Peverell et al (2005) and also found there during this current study. Notwithstanding a number of intensive surveys (Hogan and Vallance 2004, Peverell et al 2005), only one *O. mossambicus* was ever captured from the Wild River downstream of the weirs and this was in an ephemeral pool (Terry Vallance DEEDI personal communication). After earlier studies had confirmed the presence of Mozambique tilapia in both Herberton Weirs, consultations were held with the community and government agencies to canvas possible eradication and/or control options. The major consideration during these meetings was that, as the weirs provided a potable water supply for the township of Herberton, any agreed eradication method needed to be safe and not threaten the reticulated water supply (Hogan and Vallance 2004). These criteria could not be met, so control measures were limited to irregular electrofishing surveys up until this study began in October 2006.

This study aimed to assess the effects and utility of a control program on an established *O. mossambicus* population in the Herberton Weir that primarily involved the routine removal of fish using electrofishing techniques.

6.2 Study site

The Herberton Weir (17°22’S, 145°25’E) are instream impoundments located on the headwaters of the Wild River, a major tributary of the Herbert River at an altitude of about 1020 m (Figure 24). The weirs provide the primary water supply for the small township of Herberton, which has a population of about 1500 people. The two weirs are about 800 m apart, with the top weir having a surface area of about7.6 ha and a maximum depth at full supply level (FSL) of 10 m (Figure 24). The bottom weir is older and has a surface area of around 2 ha and a maximum depth at FSL of 8.3 m, although the average depth is much shallower. Water from the upstream weir is used to top up the bottom weir by gravity feed, which is directed from there into the town water reticulation system. The catchment is largely undisturbed, comprised of *Eucalyptus resinifera*, *Eucalyptus portuensis* and *Syncarpia glomulifera* (Turpentine) medium schlerophyll woodland with mesic associates (Tracey and Webb 1975). The littoral zones of both weirs have discontinuous bands of reeds (eg *Eleocharis* sp) while the open water macrophytes include *Nymphea indica*, *Vallisneria* sp and *Utricularia* sp (Figure 25). In the top weir, Hobo® pendant temperature/light data loggers, programmed to record at hourly intervals, were deployed at surface (0.5 m depth) and bottom (8.0 m depth) stations from January 2007.

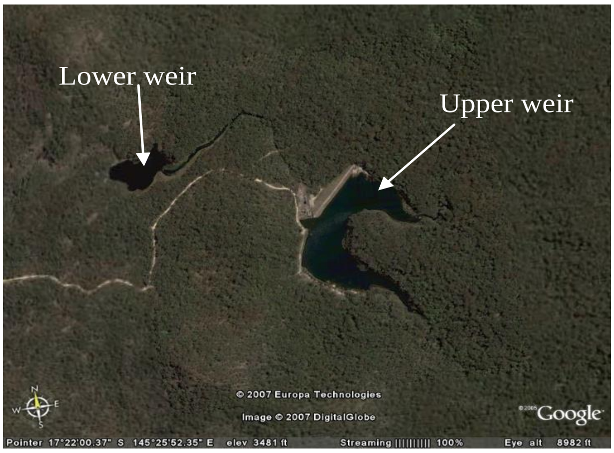


Figure 24: Satellite image showing the relative location of the two Herberton Weirs.

Figure 25: Top Herberton weir. Note reeds in the foreground.

6.3 Methods

Fish were caught in the top weir on a monthly basis from October 2006 until July 2009 using a boat-mounted Smith–Root 7.5 GPP electrofisher and two netters. Further details of the techniques used for catching these fish are given in the General Methods section of this report. Initially the electrofisher boat was driven around the perimeter of the weir until 50 adult *O. mossambicus* were caught. Electrofishing 'on-time' was recorded as a measure of effort. From May 2007 a scarcity of fish resulted in all fish, regardless of size, being sampled, and sampling also extended to the bottom weir to increase catch numbers. The target sample size remained at 50 fish (all size classes) and electrofishing on-time was recorded as a measure of effort. All fish collected throughout this study (October 2006–July 2009) were weighed to the nearest 0.1 g and measured to the nearest 1 mm. In the laboratory, otoliths were removed from fish >100 mm TL for sectioning and ageing. Gonads were also removed, weighed to the nearest 0.01 g and visually rated for maturity (see Reproductive Strategies section for further details).

Some historical data collected by Hogan and Vallance (2004) has also been included in this chapter. They also used a boat-mounted Smith-Root 7.5 GPP electrofisher to collect length and weight data on *O. mossambicus* but did not record electrofishing on-time or any other measure of fishing effort.

6.4 Results

6.4.1 Sampling between 2003 and 2004

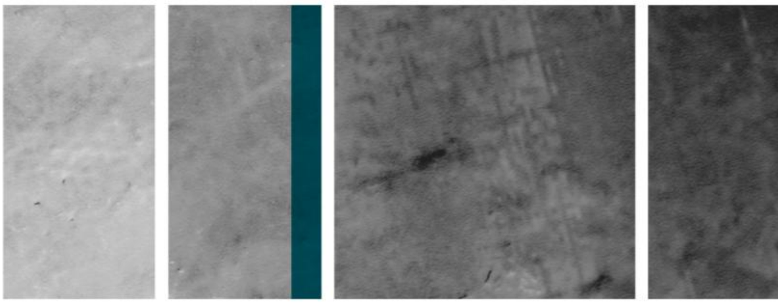
There was limited information available on Mozambique tilapia populations in the top Herberton Weir before 2006 and most of this information is contained in a report by Hogan and Vallance (2004). In November and December 2003, they removed 90 adult fish through electrofishing and a further 4523 fish in 2004 (Terry Vallance DEEDI personal communication). No information was recorded on the electrofishing effort required to catch these fish, but about 93% of *O. mossambicus* in a sample of 991 captured in February and early March 2004 were less than 50 mm. This suggests that at the time, juveniles dominated the population. No Mozambique tilapia were found by Hogan and Vallance during sampling in the bottom weir.

6.4.2 Sampling between 2005 and 2009

Regular monthly electrofishing sampling in the top Herberton Weir occurred between October 2006 and July 2009 and 1668 fish were caught. Before this sampling, 21 fish were caught in initial reconnaissance electrofishing of the weir in October and December 2005, but these have not been included in the major analyses. From May 2007, poor catches in the top weir meant the monthly sampling was extended to include the bottom weir to obtain sufficient samples for the reproductive studies. From May 2007 to July 2009 only small numbers (n=230) of *O. mossambicus* were caught in the bottom weir.

6.4.3 Changes in *O. mossambicus* abundance and population size structure in the top weir

The population size structure and catchability (catch per unit effort, CPUE) of *O. mossambicus* in the top weir changed dramatically over the study period from October



2006 to June 2009. The mean size of fish caught during monthly sampling progressively declined over the entire period (Figure 26).

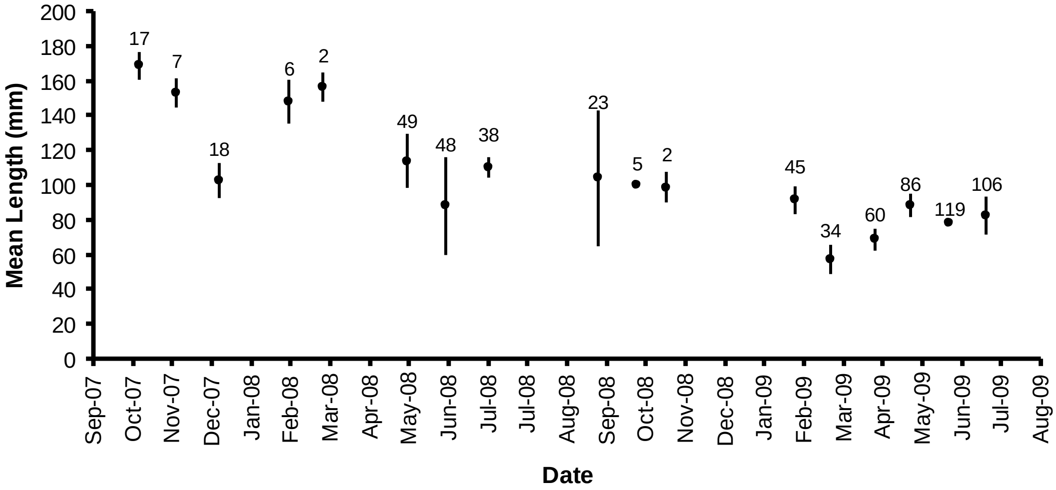


Figure 26: Mean monthly lengths (mm) of *O. mossambicus* caught in the top Herberton weir between October 2006 and July 2009. Total monthly sample sizes are given above each data point.

Figure 27 shows the monthly ratio of the number of immature to mature fish sampled in the Herberton Weirs. Up until early March 2008, the ratio was generally low (about < 1), indicating a relative high abundance of the mature fish present in the weir. After this, and with the exception of months where the sample sizes were small, immature fish increasingly dominated monthly catches.

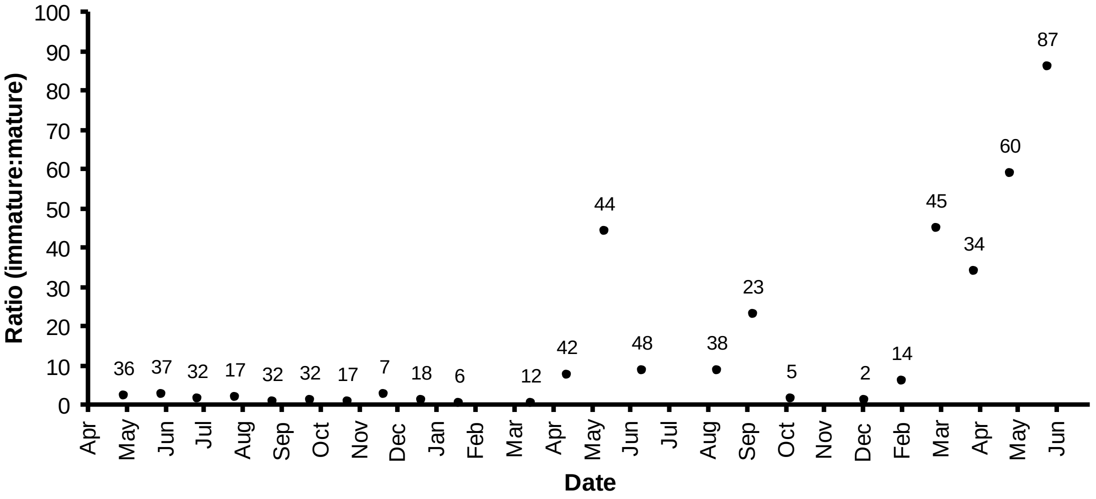


Figure 27: The ratio of immature to mature *O. mossambicus* caught in Herberton weirs between May 2007 and July 2009. Total monthly sample sizes are given above each data point.

Figure 28 shows a progressive decline in the number of mature fish caught. The maximum monthly CPUE was over 0.5 fish caught per minute of electrofishing time in February 2007, but after about June 2008 the monthly CPUE had dropped to levels of consistently <0.1 fish caught every minute of electrofishing time.

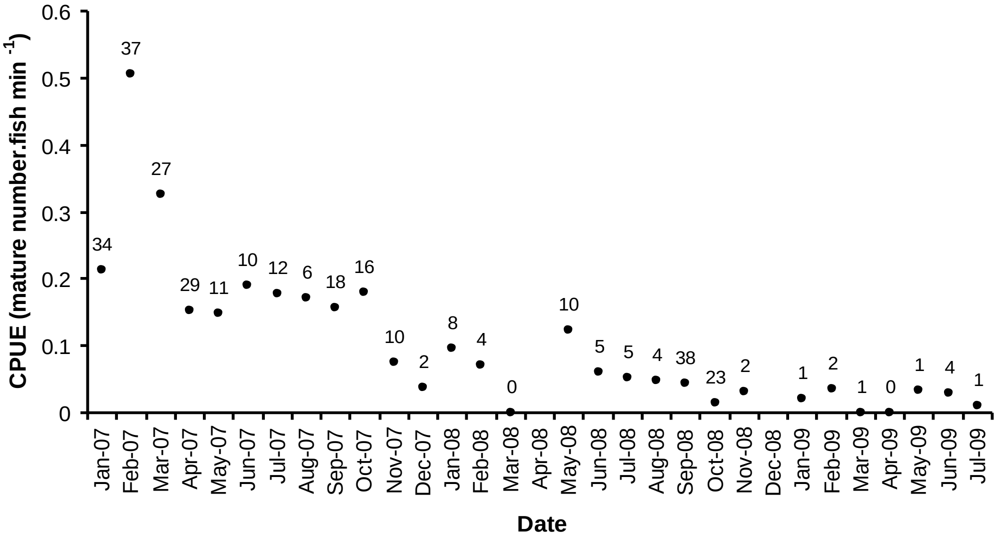
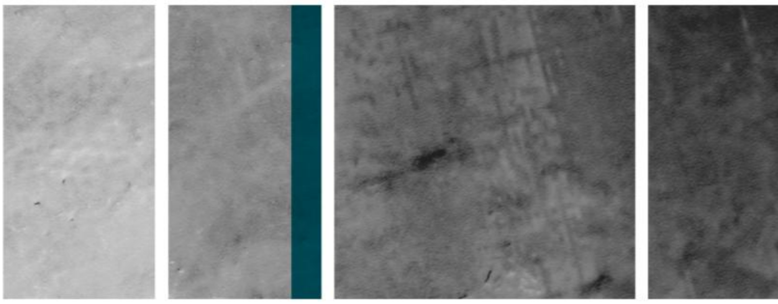


Figure 28: Mean monthly CPUE for *O. mossambicus* caught in the top Herberton Weir between November 2006 and July 2009. Total monthly sample sizes are given above each data point.

The size–frequency of *O. mossambicus* sampled from the top Herberton Weir across the duration of the study is shown by year in Figure 29. Over the four years, the size structure of the population changed, with fish <100 mm TL dominating samples during 2008 and 2009. In 2009, relatively few fish over 100 mm were captured, whereas in 2006 and 2007 most of the fish caught were in the larger size classes. Before May 2007, larger fish were selectively targeted and this may explain the lack of juveniles present in earlier samples, in particular during 2006. The larger size classes dominated the electrofishing catches in 2007, but this trend was reversed in 2008 with the recruitment of a large cohort of 0+ age class fish into the top Herberton weir (Figure 30). In 2009, the available six months of data show that 0+ fish also dominate the catches obtained for this year.



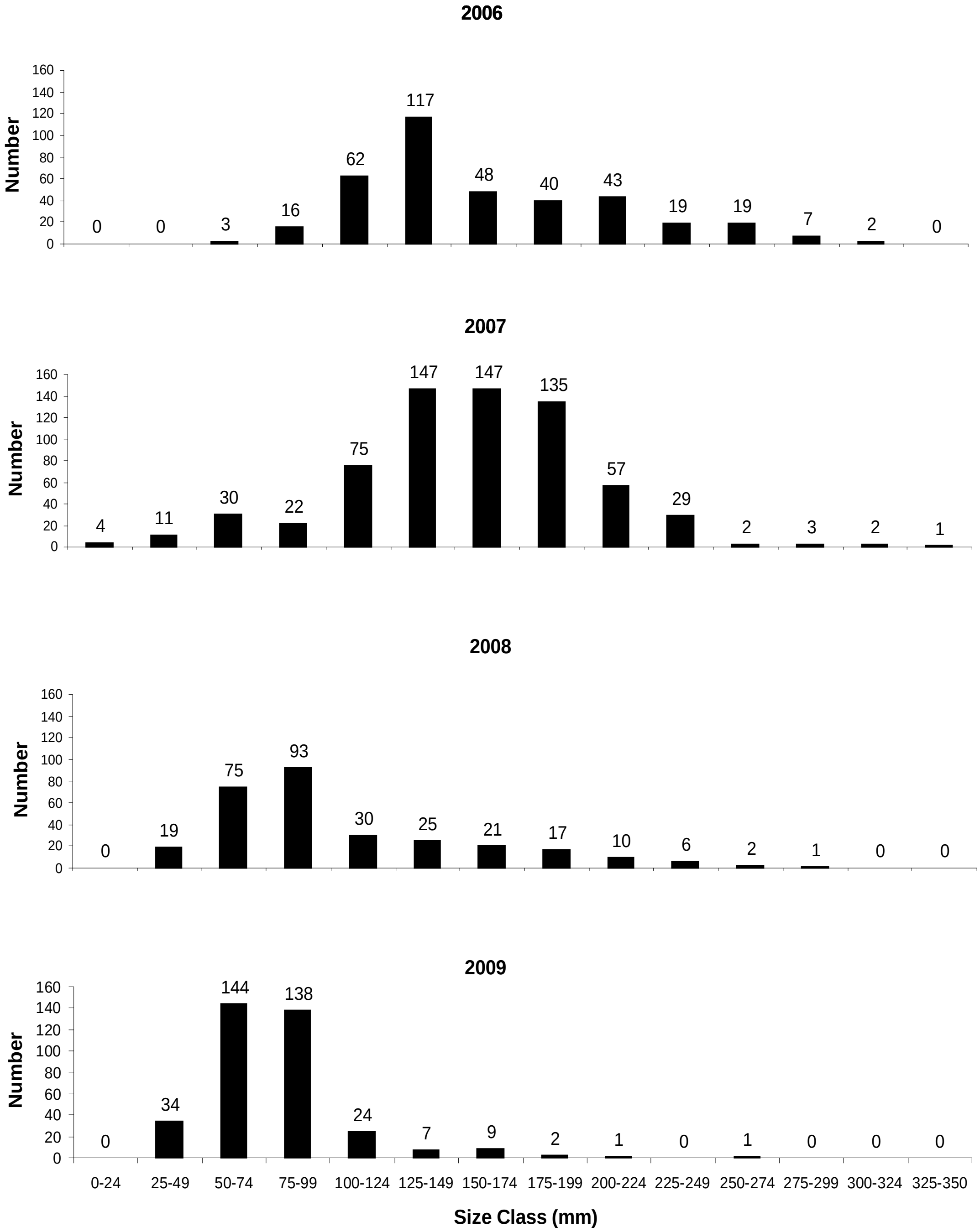


Figure 29: Size-frequency of Mozambique tilapia caught from the top Herberton Weir from October 2006 to July 2009.

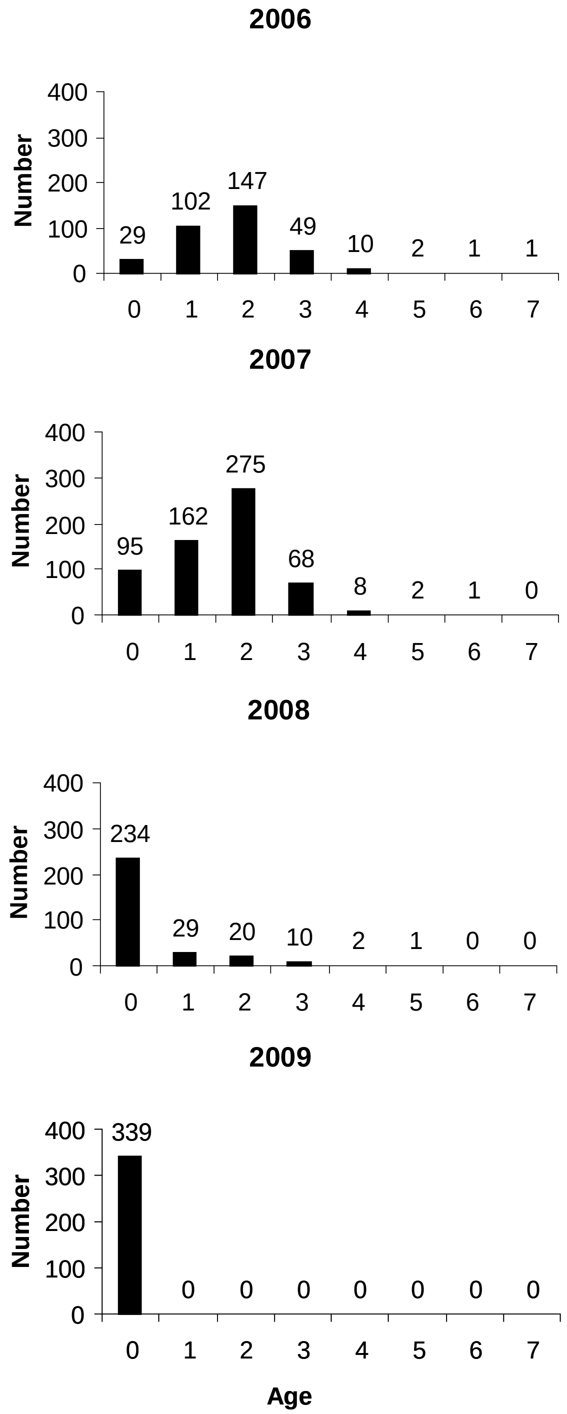
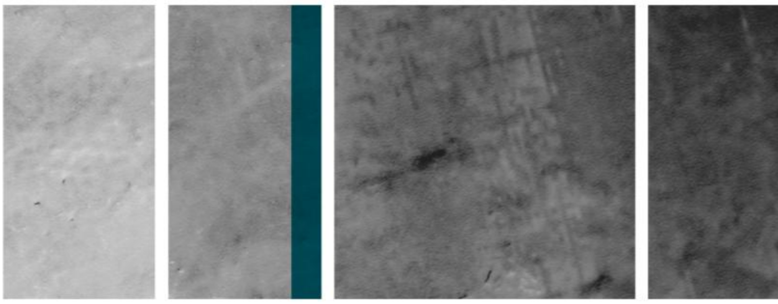


Figure 30: Age-frequency of Mozambique tilapia caught in the top Herberton Weir from October 2006 to July 2009. Data is incomplete for 2009, with further samples to be aged.



6.4.4 Changes in *O. mossambicus* abundance and population size structure in the bottom weir

From May 2007, *O. mossambicus* samples were also collected from the bottom weir. As found in the top weir, larger size classes (>100 mm TL) dominated catches during 2007 (Figure 31), but monthly electrofishing appeared to quickly remove these bigger fish. Fish collected during 2008 and the first seven months of 2009 were mostly juveniles (<100 mm TL).

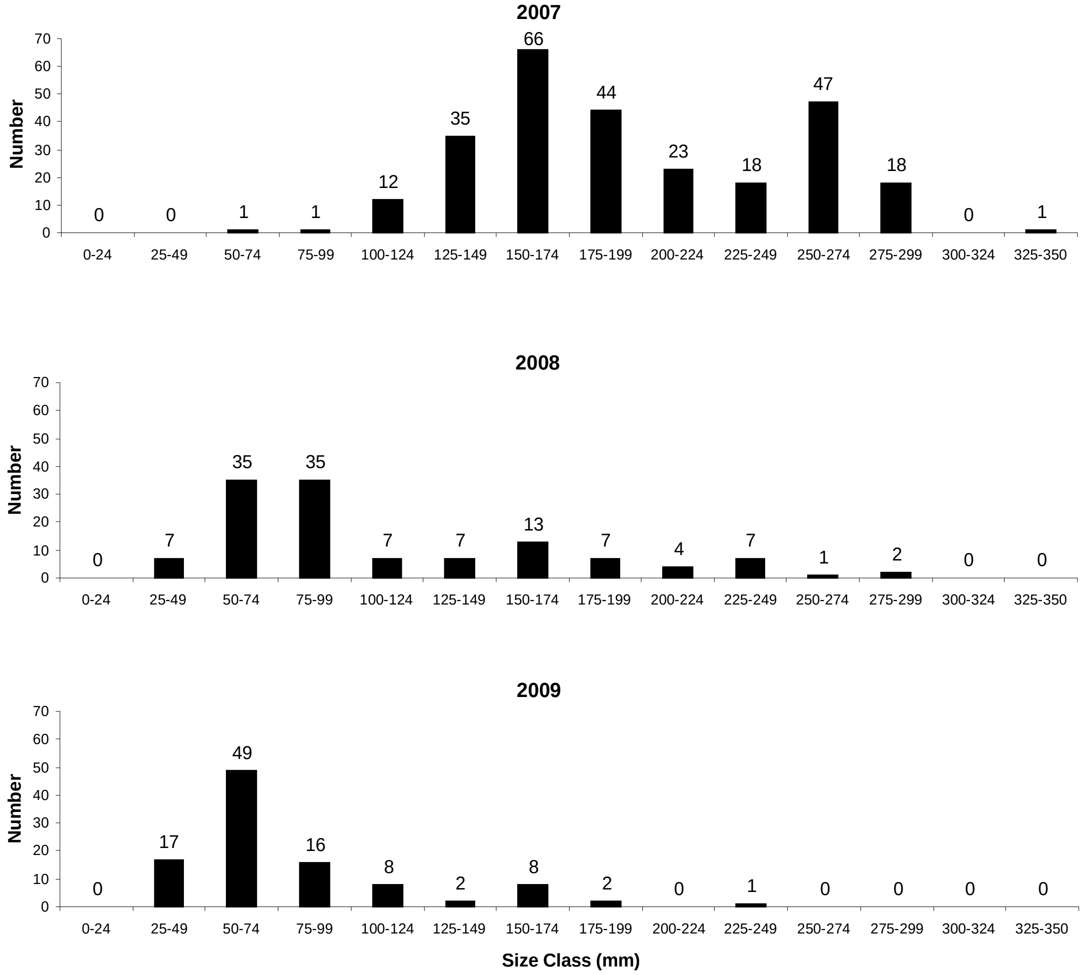


Figure 31: Size-frequency of Mozambique tilapia caught in the bottom Herberton Weir between May 2007 and July 2009.

6.4.5 Predation and other control strategies

During monthly sampling, *L. unicolor* was observed to be preying on Mozambique tilapia eggs and larvae, apparently disgorged from brooding fish as a result of electrofishing activities. This observation added support for the use of biocontrol agents such as introduced predators as part of an integrated management plan for Mozambique tilapia control. Monofilament gill nets were also trialled in the top weir during daylight in April 2009, but caught no Mozambique tilapia.

6.5 Discussion

The commencement of routine monthly electrofishing activities in late 2006 appears to have resulted in a progressive decrease in the relative abundance of older, mature Mozambique tilapia resident in the top Herberton weir. At the same time, there was a corresponding increase in the numbers of juvenile fish, particularly 0+ aged fish, caught in the top weir. Similar changes in the size structure of *O. mossambicus* were observed in the bottom weir after sampling began there in May 2007. Given that an alternative sampling methodology using gill nets failed to catch any fish, it would appear likely that the decrease in the catch of older larger fish was due to a reduction of their abundance rather than through gear avoidance.

The limited available data on the size structure of Mozambique tilapia populations in the top weir in 2003 and 2004 suggested that the population at that time was dominated by juveniles less than 50 mm TL (Hogan and Vallance 2004), but this had changed by late 2006 when this study began. Unfortunately, no records are available documenting the timing of the introduction of *O. mossambicus* into the Herberton Weir system, but it may have been that the apparent dominance of juveniles in 2003–2004 was because Mozambique tilapia had only recently colonised these impoundments and the population at that time was in a rapid growth phase (Golani 1999). The changes in the monthly ratios of immature and mature fish presented above suggest both good recruitment from the summer spawning season of 2008/09 and that the numbers of mature adults during this period were low. This low catch rate may be a result of mature fish learning to avoid being electrofished, although this seems unlikely given the failure of alternative techniques (eg spot gill netting sampling in April 2009) to catch large numbers of fish. Studies in Australia and overseas indicate that tilapia are sometimes carnivorous and even cannibalistic, particularly in an aquaculture situation (Watanabe et al 2006, Doupé et al 2009). If larger *O. mossambicus* in the Herberton Weirs are cannibalising juveniles, then the removal of larger fish through fishing operations could potentially provide a trigger for an increase in the abundance of juveniles. Lorenzen (2000) showed that recruitment in tilapia can be very strongly density dependent, with very low recruitment when stock levels are high. In reporting the results of a long-term laboratory experiment with *O. mossambicus*, Silliman (1975) noted that there was clear evidence for density dependence in the level of recruitment as well as the somatic growth of adults.

Introduction of a predator can potentially be an effective means of control. However, in hindsight, *L. unicolor* might not be the best species to use to control Mozambique tilapia in the top Herberton Weir. Arthington et al (1984) noted that *L. unicolor* and eels coexisted with *O. mossambicus* in a southeastern Queensland impoundment, where they still exist today. Pusey et al (2004) noted that piscivory was important to the diet of *L. unicolor* in impoundments with their results suggesting that, because of gape limitations, piscivory is greatest in fish >80 mm standard length. In 2005, Pearce and Kistle (2005) measured a sample (n=25) of newly established stock of *L. unicolor* from the top weir and only seven were longer than 80 mm TL. Even though *L. unicolor* will opportunistically feed on the early life stages of *O. mossambicus*, the spangled perch present in the weir system are relatively small and, because of parental care, the window of opportunity for them to prey on juvenile Mozambique tilapia may be relatively short. A remedial measure may be to move larger spangled perch from the bottom weir into the top weir, although their limited size (up to about 150 mm TL) and the fast growth rate of *O. mossambicus* suggests that this strategy might not be particularly effective.

Another option could be the introduction of other predators, although currently there is scant evidence of other native species selectively preying on *O. mossambicus*. Webb (2007) found remnants of *O. mossambicus* in the stomach contents of *Glossamia aprion* and *Strongylura krefftii* (but not in *L. unicolor*). It may be better to examine the possibility of sequentially stocking small numbers of juveniles of other native apex predators — *Lates calcarifer* (barramundi) for example — over a number of years. Barramundi has a catholic diet (Davis 1985) and in the absence of other suitable prey (as is the case in the Herberton Weirs), would undoubtedly feed on Mozambique tilapia. Sequentially stocking *L. calcarifer* has a number of advantages: the species is readily available from local hatcheries, the stocking is eventually reversible as this species needs saltwater to spawn (Mackinnon and Cooper 1987) and the individuals released into the impoundment will eventually try to move downstream. Stocking an apex predator such as *L. calcarifer* could have an adverse impact on the few native fish species resident in the weir, although both species richness (n=4) and the relative abundances of the other species are low. None of the native species present in either of the weirs are of conservation significance. A disadvantage to using *L. calcarifer* in the Herberton Weir system for predator control is that it might not be able to easily tolerate the relatively low winter temperatures. The species is thermophillic, requiring water temperatures higher than 22–25°C for growth and reproduction. In 2009, a large number of *L. calcarifer* deaths were recorded in southeast Queensland impoundments when water temperatures fell to 11.5°C (Sawynok 2009). In the Herberton Weirs, water temperatures during winter average 18°C and 16.5°C at 0.5 m and 8 m below the surface respectively — well below *L. calcarifer*'s preferred range. Given that the weirs are also domestic water supplies, it would be prudent to initially stock only small numbers of fish to avoid the possibility of a large fish kill affecting the water quality of the weirs.

A pivotal concern is that, if left unchecked, the population of *O. mossambicus* in the Herberton Weirs could colonise downstream reaches of the Wild River and eventually spread throughout the Herbert River catchment. Downstream areas have been surveyed on a number of occasions since Mozambique tilapia were first discovered in the weirs (Hogan and Vallance 2004, Peverell et al 2005) and only one Mozambique tilapia has been found, in an ephemeral waterhole immediately downstream (Terry Vallance, DEEDI personal communication). Given the right conditions, *O. mossambicus* has the capability of colonising more permanent sections of the river further downstream. Control programs would be required to limit range expansion of this pest species.

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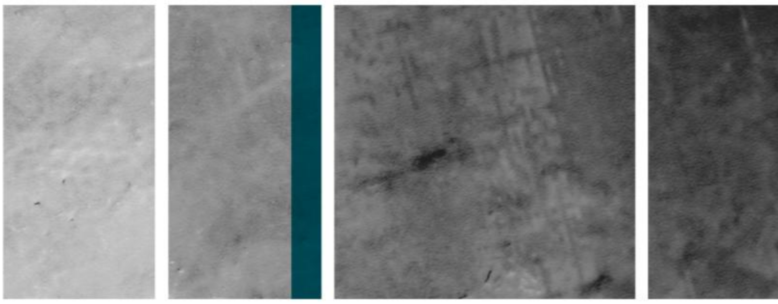

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7 Conclusions

Feral populations of two species of tilapiine fishes, *O. mossambicus* and *T. mariae*, are now well established in a variety of freshwater habitats in northeastern Australia. These habitats include coastal streams, impoundments, freshwater drains and ornamental ponds. The successful colonisation of these varied habitats is in part due to the plastic nature of tilapia's life-history characteristics, including their reproductive biology and growth parameters. This plasticity was evident from feral populations of both *O. mossambicus* and *T. mariae* demonstrating a remarkable capacity to successfully invade new and sometimes marginal habitats. Examples of marginal habitats successfully colonised by these species included shallow freshwater drains, eutrophic urban ponds and cooler, higher-altitude impoundments. Remarkably, all of these locations were situated within a narrow 67 km radius. Life-history adaptations that were found to contribute to such successful colonisations included a decreased size- and age-at-first maturity, reduced somatic growth intervals and higher relative fecundities.

The successful invasion of a drain at Kewarra Beach by *O. mossambicus* highlighted the ability of this species to adapt to very harsh conditions. The environment at this site was characterised by poor water quality due to urban runoff, high summer water temperatures, dramatic water level drops during the drier months, and very poor instream habitat dominated by exotic grasses with no riparian zone. *O. mossambicus* adapted to these conditions by maturing earlier and at a smaller size and by increasing their mean relative fecundity. The ages at maturity for male and female fish from the Kewarra Beach site were approximately five times less than the values for fish obtained from the upland Tinaroo Falls Dam population. Similarly, there were also considerable differences in the male and female sizes at maturity between sites, with *O. mossambicus* resident in Tinaroo Falls Dam having values of more than three times those estimated for fish from the Kewarra Beach Drain site.

Monthly gonad somatic indices and monthly gonad developmental stages suggested that in coastal areas, *O. mossambicus* spawns throughout most of the year. In cooler, higher-altitude impoundments such as Tinaroo Falls Dam and the Herberton Weirs, spawning of *O. mossambicus* appears to peak during the warmer, summer months. Males of both species of tilapia were found to be larger than the females, with males dominating the larger size classes of fish. A high level of parental care (ensuring the majority of eggs and larvae survive to juvenile stages) and the ability to spawn multiple broods throughout a year-round reproductive season was also found to give both species of tilapia a significant competitive advantage over native fishes.

The CARPSIM model was successfully adapted for use with *O. mossambicus* and *T. mariae*. Some of the modelling suggested that pseudo-extinction could be achieved using very high, continuous levels of fishing pressure or a combination of fishing pressure and recruitment failure. The level of intervention/s used in the simulation scenarios where pseudo-extinction was achieved was extraordinary high. At lower values, modelling indicated that these control measures could drive populations down to low base levels, but recovery followed relatively quickly after intervention ceased. This appeared to be the case in the Herberton Weir population, where both fishing pressure and predation were used as the primary control techniques. These interventions first resulted in a drop in abundance of *O. mossambicus* before the population stabilised at a lower fish density. Some authors have suggested that these control techniques might be more suited to slower-growing long-lived species whereas faster-growing short-lived species such as tilapia might respond better to spawning sabotage or male-dominance approaches. It is possible that gene modification technology currently under development by CSIRO for use on carp could also be used for tilapia control. In anticipation of this, it would be useful to develop the CARPSIM software further to allow for the simulation of male-dominance scenarios within tilapia populations. However, in the absence of more effective methods (such as genetic modification), removal-type interventions remain the primary control measures available for tilapia population management.

Models such as CARPSIM have proven to be extraordinarily useful as education and extension tools. Simulation models showing the effects of various management interventions on tilapia have already been demonstrated to state government fisheries managers, with feedback from this exercise being very positive. It is planned to continue to build on this exercise through a series of in-house seminars to further demonstrate the utility of the CARPSIM model to fisheries managers, research staff and interested community groups. This will give these stakeholders an insight into the likely short-, medium- and long-term impacts that the various management interventions will have on feral populations of tilapia and help them to decide the most appropriate control measure for specific situations.

The control experiment conducted at Herberton Weir provided a series of valuable lessons on both the mechanics and complexity of feral fish population control operations. When the *O. mossambicus* population was first sampled in 2003, it was apparently newly established and in an expansion phase with an abundance of juvenile fish. The addition of a predator (*L. unicolor*) in an attempt to reduce the number of juveniles might have had some influence, as sampling in late 2006 found the population to be dominated by larger size classes of individuals. Monthly research electrofishing from October 2006 to mid-2009 appears to have resulted in a progressive decrease in the relative abundance of older and larger fish present in the top weir site. At the same time, there was a corresponding increase in the number of juvenile fish, particularly 0+ fish caught during sampling events. The initial reduction in juvenile numbers might not have been solely due to predation by *L. unicolor*, but could have been influenced by other factors such as density dependence, or cannibalism of young by larger Mozambique tilapia. Ongoing removal of fish through monthly electrofishing operations appears to be maintaining the adult tilapia population at a relatively low abundance. The challenge now is to devise and implement an effective strategy to eliminate the remaining fish from the weirs.

There is little doubt that the spread of feral tilapia in Australia will continue through both natural and anthropogenic pathways. The population age structure of *T. mariae* in Tinaroo Falls Dam suggests that this population was established comparatively recently, probably as a result of downstream movement of fish from farm dams and creeks in the upper Barron River catchment. The recent and well-publicised operation to eradicate anthropogenically introduced *T. mariae* and *O. mossambicus* from Eureka Creek in the Gulf of Carpentaria drainage is a stark reminder of the willingness of some community members to actively spread pest fish to new areas. While there are hefty penalties in place to deter the translocation of pest fish, there is a continuing need to address this problem through targeted education programs, community engagement and research.

7.1 Future research directions

Further tilapia research opportunities that the Invasive Animals Cooperative Research Centre or other organisations might wish to consider funding include:

1. Continuing with the control experiment at Herberton Weir by:
   - assessing the impact of the introduction of another predator to this system (eg barramundi *Lates calcarifer* or a similar species)
   - approaching the Tablelands Regional Council to drop the water level of the top Herberton Weir during the breeding season to deny juvenile *O. mossambicus* access to nursery areas, while simultaneously increasing the intensity and frequency of electrofishing fish-down operations
   - increasing the intensity and frequency of fish-down operations
   - using other fishing methods, as well as electrofishing, in the fish-down operations
   - investigating aggregations and schooling of tilapia and the feasibility of implementing a 'Judas' fish strategy using radio-tagged individuals.

2. Using acoustic and radio-tagging technology to investigate:
   - habitat preferences of *O. mossambicus* resident in tropical impoundments (eg Herberton Weir) — his could be done in conjunction with a Judas fish strategy
   - home range and short- and medium-term mobility of feral *T. mariae* in a tropical river system — this could involve setting up an acoustic array in the lower Mulgrave River to determine movements of *T. mariae* in that river system.

3. Continuing to investigate the genetic population structure of *O. mossambicus* in Australia by:
   - identifying pathways of spread across the landscape
   - identifying the presence of hybrid strains within Australia
   - investigating the likely source/s of origin for the introduction of Mozambique tilapia into Australia
   - mapping the genetic population structure of Mozambique tilapia within Australia, including potential management units and barriers to gene flow
   - identifying genetic factors that contribute to the success of tilapia as an alien species.

4. Further developing the CARPSIM model to simulate other types of interventions for tilapia (eg the addition of more than one predator), the use of a wider range of integrated control measures, and the inclusion of male-dominance strategies to reduce population numbers.
5. Investigating the use of pathogens to control tilapia populations in Australia. Some recent Israeli work has reportedly identified the presence of a tilapia larvae herpesvirus (TLHV) that can cause up to 90% larval mortality in different tilapiine species. While there does not appear to be any existing cultures of this virus in Israel (Professor R Avtalion personal communication), it or other tilapia-specific pathogens could have been isolated in other facilities. This issue could be progressed through an intensive literature search to

determine the existence of potential tilapia-specific pathogens, followed by contacts with overseas workers and, eventually, a study tour to confirm the presence and availability of cultures.

6. Examining the feasibility of sex-biased control strategies using chemical and/or genetic technologies.