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An evaluation of electrofishing as a control measure for an invasive tilapia (*Oreochromis mossambicus*) population in northern Australia

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Abstract. Combating the spread of invasive fish is problematic, with eradication rarely possible and control options varying enormously in their effectiveness. In two small impoundments in north-eastern Australia, an electrofishing removal program was conducted to control an invasive tilapia population. We hypothesised that electrofishing would reduce the population density of *Oreochromis mossambicus* (Mozambique tilapia), to limit the risk of downstream spread into areas of high conservation value. We sampled by electrofishing monthly for 33 months. Over this period, there was an 87% decline in catch per unit effort (CPUE) of mature fish, coupled with a corresponding increase of 366% in the number of juveniles, suggesting a density-dependent response in the stock–recruitment relationship for the population. Temperature was inversely related to CPUE ($r = 0.43$, lag = 10 days), implying greater electrofishing efficiency in cooler months. The reduction in breeding stock is likely to reduce the risk of spread and render the population vulnerable to other control measures such as netting and/or biological control. Importantly, the current study suggests routine electrofishing may be a useful control tool for invasive fish in small impoundments when the use of more destructive techniques, such as piscicides, is untenable.

Additional keywords: alien species, introduced fish, non-native fish, population control, Wet Tropics.

Introduction

The spread of non-native fish into novel habitats is a rapidly accelerating global trend, resulting in the homogenisation of faunas (Olden 2006). Humans have facilitated the movement of fish species outside their natural ranges for food aquaculture, ornamental fish, and recreational and commercial fisheries (Gozlan 2008). Many of these introductions have resulted in self-sustaining populations, with some deleterious impacts on aquatic communities. For example, non-native species have been implicated as a causal factor in 68% of North American fish extinctions (Miller *et al.* 1989) and in Australia, the spread of non-native fish is considered a major factor in the selection of threatened status for 42% of native fish species of conservation concern (Lintermans 2004).

Unfortunately, the control of non-native fish is difficult to achieve owing to their ability to spread quickly and the three-dimensional complexity of aquatic habitat. Mack *et al.* (2000) listed three general strategies available to combat the threat of invasions: (1) prevention, (2) eradication before widespread establishment, and (3) low-level population maintenance. Prevention is the most effective management strategy in terms of avoiding the establishment of non-native fish, but is not always possible, especially when deliberate introductions of aquatic organisms can account for half of all problem invasions (Mack and Erneberg 2002). Eradication of non-native fish traditionally entails physical removal through the use of piscicides (Meronek *et al.* 1996). However, the effectiveness of chemicals is generally limited to small, contained systems (Lazur *et al.* 2006;

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Britton *et al.* 2009), where impacts on water quality and/or non-target species are of minimal concern (McClay 2000). When the use of piscicides is untenable, control mechanisms that act to maintain invasive populations at low levels, thereby limiting their impacts on recipient ecosystems, may be the best option (Myers *et al.* 2000). Some more common methods include *mechanical control* techniques (traps, nets, line fishing, electrofishing), which, unlike chemical applications, allow higher target specificity (Knapp and Matthews 1998). However these techniques often require intensive application, both temporally and spatially, to keep populations suppressed. Invasive populations may also be *biologically controlled*, through the introduction of a competitor, predator, parasite and/or pathogen. Biological methods are often attractive to resource managers as unlike other control methods they do not generally require intensive and ongoing interventions (Secord 2003). However, such introductions for control purposes can result in irreversible ecological consequences on non-target species and ecosystem function (Howarth 1991). Because highly modified habitats are often implicated as a factor in the success of invasive species (Moyle and Nichols 1974), *habitat restoration* of aquatic ecosystems may also assist with the control of invasions and reduce impacts on native species (Scoppettone *et al.* 2005). However, this method may require a long-term management commitment and is often expensive to implement. Further, the invasiveness of some species may not be related to habitat integrity (Smokorowski *et al.* 1998).

Oreochromis mossambicus (Mozambique tilapia) is listed in the Global Invasive Species Database (http://www.issg.org/database) as being in the top 100 invasive alien species on the planet. Pérez *et al.* (2006) described *O. mossambicus* as a 'model invader' owing to key biological characteristics such as tolerance to wide-ranging ecological conditions, generalist dietary requirements, rapid reproduction with maternal care, and the ability to compete with native fish through aggressive behaviour. Given suitable environmental conditions, these fish have become successfully established in almost every region in which they have been cultured or imported (Costa-Pierce 2003). Members of the *Oreochromis* species complex are regularly implicated in the decline of indigenous fishes (Pérez *et al.* 2003; Jenkins *et al.* 2010), through competitive displacement (Doupé *et al.* 2009) and habitat alteration (Starling *et al.* 2002).

In Australia, *O. mossambicus* populations have progressively spread into new catchments since their introduction in the late 1970s (Arthington *et al.* 1984). Populations are now well established in eastern Queensland drainages (e.g. Russell *et al.* 2003; Webb 2007) and the Pilbara region of Western Australia (Morgan *et al.* 2004). Chemical eradication has been trialled on small *O. mossambicus* populations in closed systems with varying success (Arthington *et al.* 1984). Effective control options are urgently needed for situations where chemical application is unsuitable, if populations are to be controlled and potential impacts on native fauna minimised. However, to date no empirical test of potential control measures other than the use of chemicals (Arthington *et al.* 1984) has been conducted on wild populations of invasive *O. mossambicus*.

In November 2003, a population of *O. mossambicus* was detected and 90 fish removed by A. Hogan and T. Vallance (unpubl. data) in a small weir in the headwaters of the Herbert River catchment in north Queensland. In early 2004, ostensibly to reduce the risk of *O. mossambicus* spreading downstream, Hogan and Vallance (2004) removed 991 mostly juvenile *O. mossambicus* from the weir over three days using electrofishing. However, no further electrofishing removals were conducted and in September 2004, *O. mossambicus* individuals had spread to a second weir at ~500 m downstream. To date, *ad hoc* surveillance surveys have found no evidence of *O. mossambicus* establishing downstream of the weirs (M. Pearce, unpubl. data). The potential for downstream movement out of the weirs is concerning, as the Herbert River catchment is the largest in the Wet Tropics bioregion of north Queensland and contains wetlands of high conservation value. Because the Herberton Weirs are a town water supply, eradication through the use of piscicides was not feasible. As an alternative, it was proposed to test the use of electrofishing as a mechanical population control to limit the probability of further spread downstream. Electrofishing removal, although at times labour-intensive, has been shown to be useful in significantly reducing the density of other invasive species (Moore *et al.* 1983; Peterson *et al.* 2008), and allows for higher target selectivity than piscicides.

The current study is the first to assess the use of electrofishing as an ongoing control tool for an invasive *O. mossambicus* population. The aim of this study was to determine if routine electrofishing could be used as a viable technique to control an invasive *O. mossambicus* population resident within the Herberton Weirs. We hypothesised that electrofishing would reduce the population density of *O. mossambicus*, to limit the risk of downstream spread into areas of high conservation value.

Methods

Study site

The Herberton Weirs (~17°22′S, 145°25′28″E) are two in-stream impoundments, which flood terrestrial habitat at full supply level (F.S.L.) and are in the headwaters of the Wild River, an upper tributary of the Herbert River in north Queensland. The weirs are the water supply for the town of Herberton, population ~1500 people, and are at an altitude of ~1000 m above sea level. The two weirs are ~800 m apart with the top weir having a surface area of ~7 ha and a maximum depth at F.S.L. of 8 m. The bottom weir has a surface area of ~4 ha and a maximum depth at F.S.L. of 6 m. Water levels remained stable over the duration of the study (2006–2009), with both weirs remaining at or very close to F.S.L. Water from the upstream weir is gravity-fed into the bottom weir and subsequently directed into a water-reticulation system. The catchment occurs in relatively undisturbed sclerophyll woodland with mesic associates (Tracey and Webb 1975). The littoral zones of both weirs have discontinuous bands of reeds (e.g. *Eleocharis* sp.), whereas open-water macrophyte beds include *Nymphoides indica*, *Vallisneria* sp. and *Utricularia* sp.

Field sampling

Electrofishing of both weirs occurred monthly for 33 months, from October 2006 until September 2009; however, catch per unit effort (CPUE) data were only collected from January 2007. Surveys were conducted with two netters and a boat-mounted

Smith-Root 7.5 GPP electrofisher (Vancouver, WA, USA) using the following settings: pulsed DC 120 Hz, 60–80% range and 1000 V. Monthly electrofishing operations involved a single-perimeter circuit of each weir with all stunned fish collected. Repeated passes of the perimeter were not carried out, as previous *ad hoc* removals before the current study indicated that very few fish were collected after the first pass (D. Russell and P. Thuesen, unpubl. data). This is likely a result of stress-induced reduction in catchability from the disturbance associated with the initial electrofishing pass (Bohlin and Cowx 1990). Fish were placed into an aerated 100-L fish bin and humanely euthanised using a lethal dose ($\sim 5\,g\,L^{-1}$) of the anaesthetic AQUI-S (Aqui-S NZ Ltd, Lower Hutt, New Zealand). Electrofishing power time-on was recorded as a measure of CPUE (fish min^{-1}). Owing to adverse weather events or gear failure, full surveys were not conducted for every month during the study period.

Water quality data

Water physico–chemical variables (dissolved oxygen, pH, conductivity, turbidity and temperature) were recorded (TPS 90FL-T model, TPS Pty Ltd, Brisbane, Australia) immediately before each sampling event. These variables were collected at a depth of ~1 m and ~4.5 m. In addition, Hobo temperature data loggers (model UA-002–64, Onset Computer Corporation Bourne, MA, USA), programmed to record at hourly intervals, were deployed at surface (1.0 m top and bottom weirs) and bottom (8.0 m for top weir only) stations from January 2007.

Laboratory analysis

In the laboratory, the total length (TL) of all fish collected were measured to the nearest 1 mm and weighed to the nearest 0.1 g. To determine fish age, the right sagittal otolith was removed and embedded in epoxy resin before thin sections (~50 μm) were cut using a Gemmasta (Shell-Lap Supplies Pty Ltd, Adelaide, South Australia) high-speed diamond saw and mounted on microscope slides. An age was assigned to each fish after counting the number of opaque bands on the otolith section under reflected light using a stereo microscope (10–40× magnification). This ageing method has been successfully used for *O. mossambicus* (Tachihara and Obara 2003) and other tilapia species (Faunce *et al.* 2002; Bwanika *et al.* 2007; Ishikawa and Tachihara 2008). Sexual maturity was assessed by inspecting all fish greater than 70 mm in TL and assigning the gonads an index of maturity using a slightly modified version of the six-point gonad-maturity classification scheme described by Davis (1982).

Data analysis

To determine if a significant relationship existed between CPUE (fish min^{-1}) and time, several regression models were fitted to the observed data and compared in order to select the model of best fit, based on the magnitude of the R^2 values and levels of significance. These models were applied to CPUE of: (a) all fish combined, (b) mature fish, and (c) immature fish.

Regression models of best fit were also applied to determine if a significant relationship existed between the water physico–chemical variables (dissolved oxygen, pH, conductivity, turbidity and temperature) at the time of sampling and CPUE. The potential for a lag period to exist in the relationship between CPUE and water physico–chemical variables was determined by cross-correlation analysis of the time series data. All analyses were carried out in GENSTAT ver. 6. (Payne *et al.* 2006).

All data collected from both weirs were combined because: (1) they exhibited the same trends in terms of population size structure, CPUE and water physico–chemical variables during the study (D. Russell and P. Thuesen, unpubl. data), (2) to increase the power of statistical analyses, and (3) both weirs are located very close to each other (~500 m) and are similar in terms of their water quality, geomorphology and hydrology.

Results

Between January 2007 and September 2009, a total of 1361 fish were caught and removed from the weirs (979 and 382 from top and bottom weirs respectively). The mean (±s.e.) monthly electrofishing power-on time for the top and bottom dams was 82.6 ± 12.5 and 37.9 ± 11.4 min, respectively.

Changes in CPUE over time

No regression model could be fitted to CPUE of all fish caught over time (Fig. 1).

Catch per unit effort of mature fish declined 87% during the study from a mean (±s.e.) of 0.19 ± 0.02 in 2007 to 0.02 ± 0.01 in 2009. An inverse exponential model best explained the relationship between CPUE of mature fish and time ($R^2 = 71.6$, $F_{2,30} = 38.82$, $P < 0.001$) (Fig. 2). Catch per unit effort of immature fish increased 366% during the study from a mean (±s.e.) of 0.18 ± 0.02 in 2007 to 0.67 ± 0.14 in 2009. However, owing to large fluctuations in CPUE, especially between winter (higher CPUE) and summer (lower CPUE) months in 2008 and 2009, no regression model could significantly explain the relationship between CPUE and time for immature fish (Fig. 3).

Relationship between CPUE and temperature

Of the water physico–chemical variables measured, only mid-day surface temperature on the day of electrofishing (recorded at 1-m depth) significantly explained the variance associated with CPUE over time. A linear model best described the relationship between CPUE and surface temperature (CPUE = 0.04–0.001 × temperature), with inverse periodicity evident between the two parameters (Fig. 1). Although this model was significant, only a small amount of residual variation was explained by the relationship ($R^2 = 0.22$, $F_{2,27} = 8.72$, $P < 0.01$). The results of the cross-correlation analyses suggest a stronger inverse relationship at a lag period of 10 days (cross correlation coefficient $r = -0.43$) (Fig. 4). Catch per unit effort of immature fish spiked substantially during the winter months in the final two years (2008 and 2009) of the study (Fig. 3).

Changes in population size and age structure

In 2007, the majority of fish caught measured between 150 mm and 225 mm (Fig. 5*a*). In 2008, the population size structure shifted, with 57% of fish caught falling between 75 and 125 mm. By 2009, fish caught in the 75–125-mm size range had increased to ~75%. In 2009, only 5 fish were collected in size classes greater than ~225 mm. The age structure of fish in the Herberton Weirs also declined over the sampling period (Fig. 5*b*). In 2007, just over half of fish caught were between

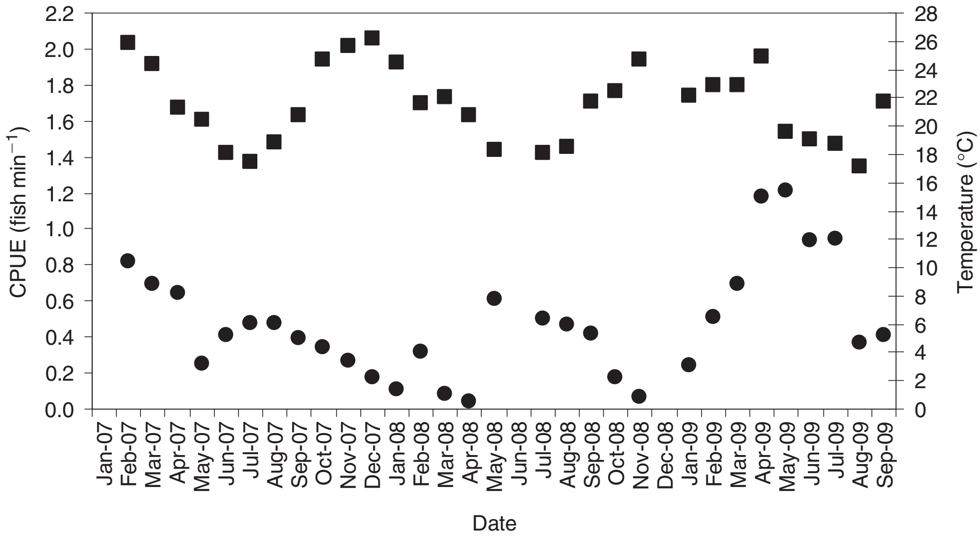

Fig. 1. Time series plot of CPUE (●) of *Oreochromis mossambicus* and midday water temperature (■).

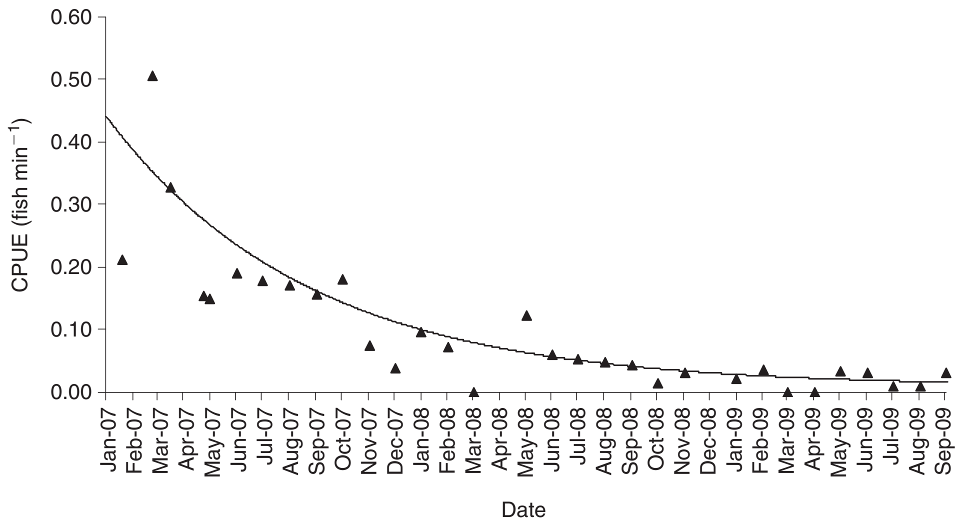

Fig. 2. Fitted relationship (inverse exponential) to CPUE of mature *Oreochromis mossambicus* and time ($R^2 = 71.6$, $F_{2,30} = 38.82$, $P < 0.001$), captured using electrofishing from the Herberton Weirs.

ages 2+ and 3+ years old, although by 2009, 95% of fish caught were less than a year old, with no fish caught greater than 3+ years old.

Discussion

Decline in CPUE of mature fish

A major finding of the current study was a substantial decline in CPUE of mature *Oreochromis mossambicus* within the Herberton Weirs as a result of electrofishing removals. We suggest that this decrease represented a real decline in the adult population for two reasons. First, recruitment in *O. mossambicus* is strongly density-dependent, with very high levels of recruitment when stock densities are low (Silliman 1975; Lorenzen 2000). During the second and third years of the current study, the CPUE of immature *O. mossambicus* substantially increased at the same time as the CPUE of mature fish declined, indicating a density-dependent response within the population. Hogan and Vallance (2004) provided further evidence of density-dependent stock recruitment within the Herberton populations. In their electrofishing surveys in 2003, Hogan and Vallance (2004) documented high juvenile and low adult densities of *O. mossambicus* (88% of fish at < 50-mm T.L.), suggesting the population was in a rapid growth phase owing to their (assumed) recent introduction to the weirs. Second, electrofishing sampling is strongly biased towards removal of larger fish (Zalewski 1983), as explained by Rushton's Law, where bigger fish are more susceptible to electrical stimulation owing

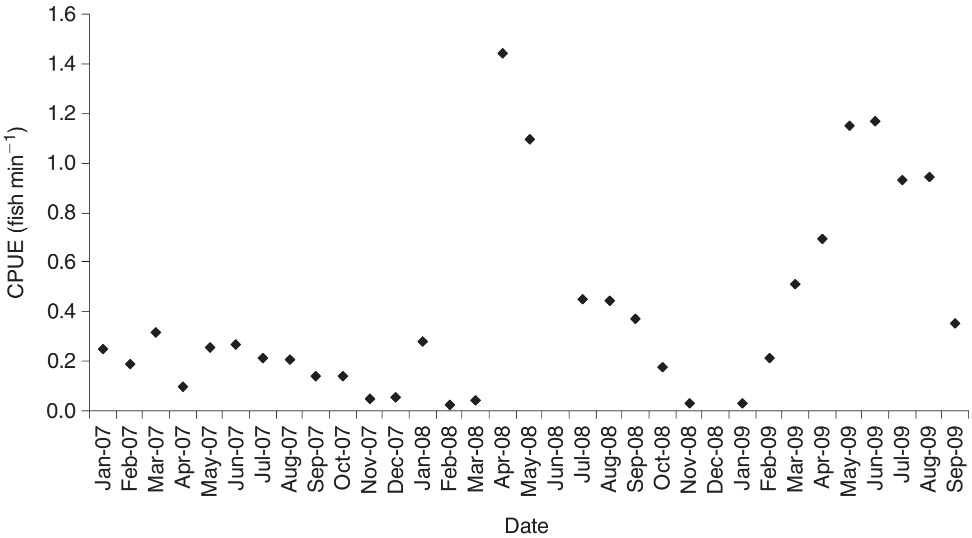

Fig. 3. Time series plot of CPUE of immature *Oreochromis mossambicus* captured using electrofishing from the Herberton Weirs.

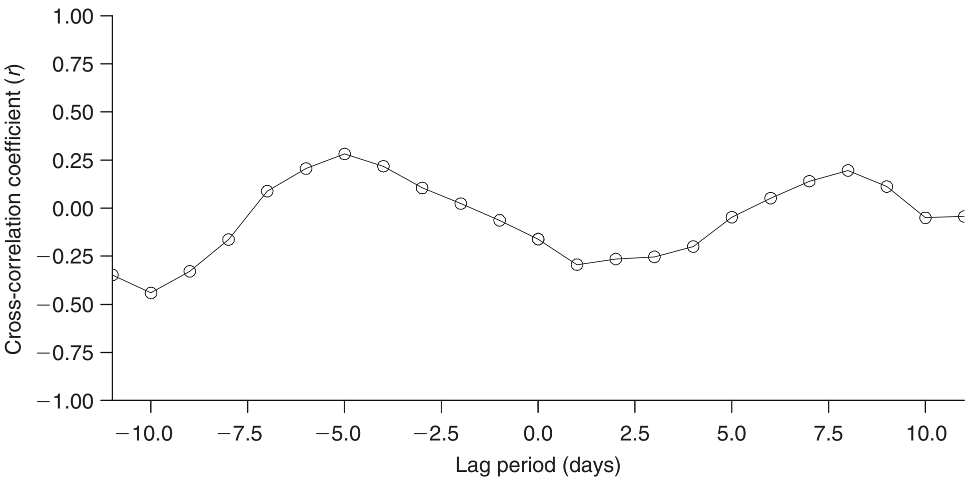

Fig. 4. Correlation coefficients of cross-correlations between CPUE of *Oreochromis mossambicus* and midday water temperature in the Herberton Weirs.

to greater total body potential and nerve length (Zalewski and Cowx 1990).

The density-dependent response exhibited by the *O. mossambicus* population in the Herberton Weirs has the potential to impair our control efforts, as the number of juveniles increase with decreasing spawning stock. This compensatory response is likely to be associated with reduced competition and cannibalism, and can be an indicator of population rebound if future electrofishing efforts are halted (Rose *et al.* 2001; Hein *et al.* 2007). However, we argue that the removal of adult fish has achieved a significant control outcome because the continuous removal of many individuals has reduced the biomass of *O. mossambicus* within the weirs to a point where the population is well below carrying capacity. Consequently, there is less pressure for individuals to disperse out of the weirs owing to limited resources. Furthermore, low spawning stock density may render the population susceptible to depensation (i.e. reduced rates of survival and reproduction that can lead to population collapse) (Myers *et al.* 1995). Although not reported for tilapia species, depensation can be triggered by a reduced probability of finding a mate, impaired group dynamics, and predator saturation (Liermann and Hilborn 2001). Finally, to date *O. mossambicus* have not become established below the weirs, assessed by downstream surveys (M. Pearce, unpubl. data). Following successful establishment, *O. mossambicus* has the ability to rapidly spread throughout a catchment (Canonico *et al.* 2005). The population has been resident in the Herberton Weirs for at least seven years. Consequently, some factor must be responsible for halting their spread, and we putatively suggest electrofishing as the most parsimonious explanation for their containment.

Variation in catchability between individuals

The inverse exponential relationship between CPUE and time (Fig. 2), coupled with the corresponding increase in juvenile

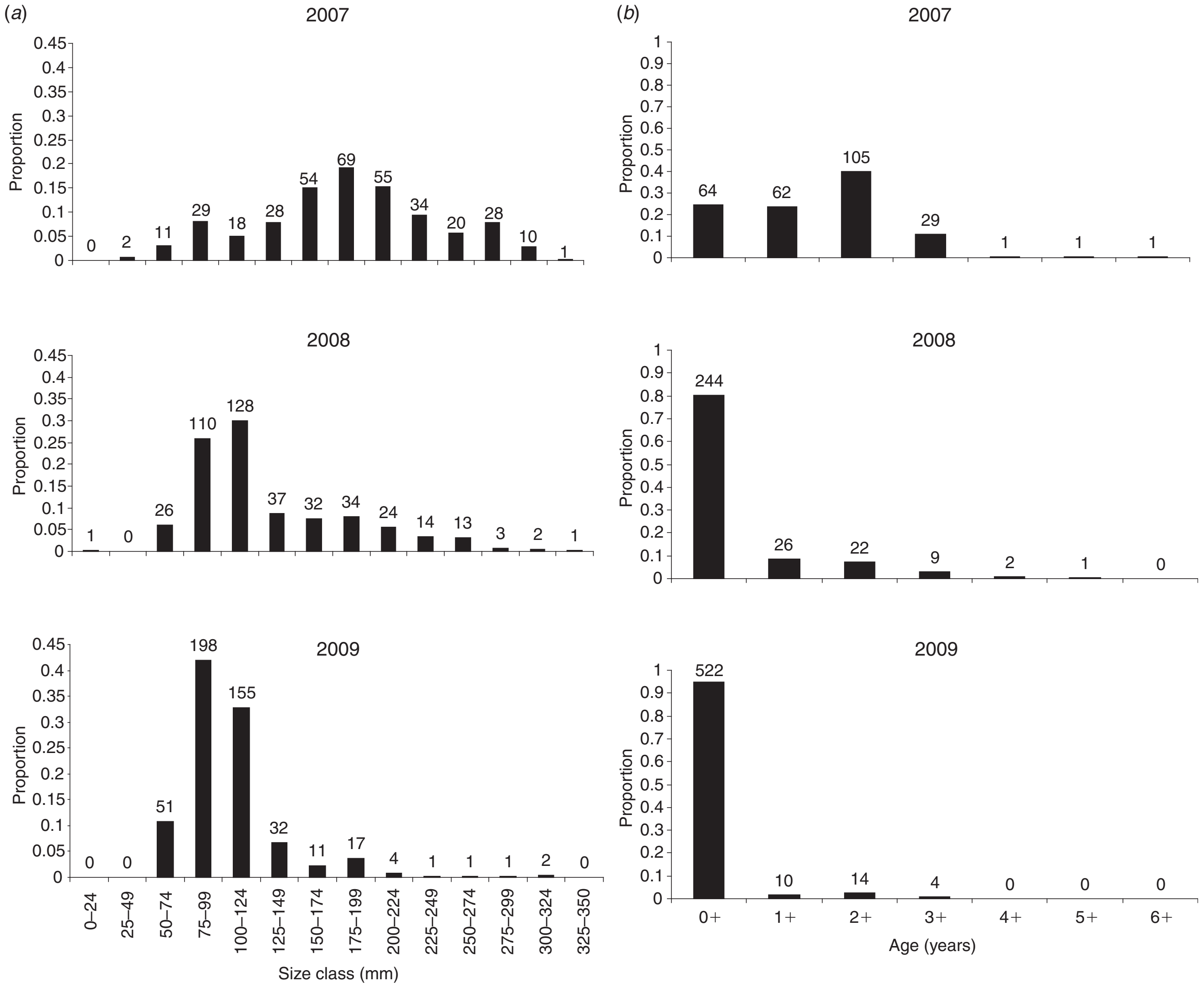

Fig. 5. Size (*a*) and age (*b*) structure of *Oreochromis mossambicus* removed from the Herberton Weirs by electrofishing between May 2007 and September 2009. Sample sizes denoted at top of columns.

numbers, suggests that electrofishing is unlikely to drive *O. mossambicus* populations in the weirs to extinction. Electrofishing estimates of salmonids have demonstrated the differential probability of capture between individuals, resulting in population underestimates of up to 25% (Bohlin and Cowx 1990). A mark–recapture experiment of *O. mossambicus* of a known population size in a control weir or weirs would be useful in determining the level of variation in catchability. Given the noxious status of *O. mossambicus* in Queensland and the limited resources available to managers to curb their spread, such a control was beyond the scope of the current study. In the Herberton Weirs, CPUE data for mature fish indicate that a crash in CPUE was achieved after one and half years of routine monthly electrofishing. In the final year of the study, the CPUE of mature fish was very low, probably because the majority of individuals with high catchability were removed from the population. Similarly, Bohlin and Cowx (1990) demonstrated that after 20 electrofishing removals of brown trout from a stream, only 3% of the population continued to evade capture.

Future control in the Herberton Weirs

To work towards full eradication, we suggest the integration of other control techniques with electrofishing to remove the few remaining individuals. Control techniques could include a netting or trapping program to target the remaining mature fish that evade electrofishing capture and/or the introduction of biocontrol such as the native piscivore, *Lates calcarifer* (barramundi), to target juveniles. Stocking of *L. calcarifer* is reversible, as this species is catadromous (MacKinnon and Cooper 1987) and can easily be fished back out of the weirs. In addition, non-target impacts are likely to be minimal as the other native fish species occurring in the weirs are not of conservation concern and occur naturally with *L. calcarifer* further downstream.

Relationship between temperature and CPUE

In the Herberton weirs, the CPUE of *O. mossambicus* was inversely related to water temperature, although it is likely that other environmental factors not measured, such as rainfall, wind, and cloud cover, also contributed to the residual variability in the relationship (Reynolds 1996). Conductivity, a significant variable affecting electrofishing efficiencies, is also affected by temperature, as water becomes more viscous and ions move less easily at lower temperatures (Cowx and Lamarque 1990). However, the very low and stable conductivities recorded in the current study ($37.7\,\mu S\,cm^{-1} \pm 1.4$ s.e.) suggest conductivity was unlikely to have contributed significantly to the variability in CPUE.

In the Herberton Weirs, winter water temperatures average 18.01°C and 16.50°C at 0.5 m and 8 m (latter depth measured in top weir only) below the surface, respectively, well below the preferred range of *O. mossambicus* (22–25°C) for optimal growth and reproduction (Chmilevskii 1998). The inverse relationship between temperature and CPUE may be explained by two non-mutually exclusive factors: (1) a reduced ability of individuals to avoid capture owing to lowered metabolism (Reynolds 1996; McInerny and Cross 2000), and (2) a higher density of *O. mossambicus* during winter months in shallow areas of the weirs with abundant macrophyte growth, to take advantage of waters warmed by solar radiation. The use of warm shallows most likely increases the rate of somatic growth when temperatures are low (Garner *et al.* 1998). Such a shift in the distribution of *O. mossambicus* to shallow habitats helps to explain a higher CPUE, as fish are more vulnerable to electrofishing in shallow waters (Reynolds 1996). A much larger spike in CPUE during winter was observed for juvenile fish, especially in the last two years of the study when their densities became higher owing to the removal of most adults with high catchability. Therefore, to limit the contribution of juveniles to successive generations, we suggest extra electrofishing effort be applied during winter months, when water temperatures drop below the thermal preference of *O. mossambicus*.

Cost of electrofishing effort

The current study suggests that with relatively little effort, the population of *O. mossambicus* in the Herberton Weirs, particularly mature individuals, can be controlled using a physical removal method. This has been achieved with an average electrofishing power on-time effort of ~120 min per month over a 33-month period. Fisheries Queensland charges a daily fee of AUD$2300 for the hire of an electrofishing vessel, three staff and a tow vehicle. Applying this rate, which may differ between State agencies within Australia and internationally, the control effort of *O. mossambicus* in the Herberton Weirs can be extrapolated to cost around AUD$27 600 annually. Lastly, if significant biomass is removed during operations, the cost of disposal will also need to be accounted for.

Conclusion

The electrofishing control program conducted in this study has demonstrated that, with relatively minimal effort, an introduced *O. mossambicus* population can be satisfactorily maintained at low adult densities in small impoundments.

Closed water bodies such as farm dams, ornamental ponds and water reservoirs are often a point source of *O. mossambicus* introduction within Australia (Arthington *et al.* 1984; Webb 2007) and other parts of the world (St Amant 1966). Consequently, if invasive fish outbreaks are detected early, electrofishing removal may prove a viable control, if the use of more destructive methods such as piscicides is not acceptable. In the current study, the primary aim of electrofishing control was to limit the risk of spread of *O. mossambicus* into a large catchment of high conservation value. However, keeping adult numbers low using this technique may be applied by managers to meet other goals such as: (a) limiting the impact of invasive fish on species of conservation concern; (b) protecting the socioeconomic value of a water body, which can be reduced owing to the proliferation of an invasive species, and (c) maintaining a low biomass within a town water supply, to mitigate the risk of contamination from a potential fish kill.

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