FRONTISPIECE

ODE TO A FERAL FISH

O fish, little fish of beautiful hue,
Did you come here by chance,
Did you come from afar?
Are there others like you from similar climes,
Beyond the horizon or round the next bend?
Are you looking for refuge, are you friend, are you foe,
Where did you come from, where will you go?
Tell me, O fish, little fish of beautiful hue,
Are you here for a while,
Or just passing through?

a.c. webb, Aug 2002

THE ECOLOGY OF INVASIONS OF NON-INDIGENOUS FRESHWATER FISHES IN NORTHERN QUEENSLAND

Thesis submitted by

ALAN CHARLES WEBB

in December 2003

For the degree of Doctor of Philosophy
in Zoology and Tropical Ecology
within the School of tropical Biology
James Cook University

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DECLARATION

I declare that this thesis is my own work and has not been submitted in any form for another degree or diploma at any other university or other institution of tertiary education. Information derived from the published or unpublished work of others has been acknowledged in the text and a list of references given.

All research procedures reported in the thesis received the approval of the JCU Experimentation Ethics Review Committee, Approval No. A444, 15 October 1997.

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ABSTRACT

This study investigated the ecology of invasions of non-indigenous freshwater fishes in northern Queensland and, in particular, examination of historical changes in their distribution patterns and establishment success in relation to their use of human altered habitats and interactions with indigenous predators and parasites.

Seventeen non-indigenous fish species were reported from northern Queensland fresh waters during the study with eleven species establishing breeding populations. Virtually all species were observed in waterways in agricultural and urban regions that have been altered by human activity. Non-indigenous fishes are continuing to disperse locally, with large-scale changes in range due to translocation by humans. Previous history of introductions elsewhere and indices of propagule pressure (frequency of introductions) and habitat matching (absolute latitudinal range overlap) are very good, though not absolute, predictors of establishment success of non-indigenous fishes in northern Queensland.

Field and laboratory studies demonstrated the importance of refuges, created by habitat disturbance, to the survivorship of some small, non-indigenous species. In the Ross River catchment, non-indigenous fishes were predominant in small streams and channels where access was blocked by dense aquatic vegetation and in isolated wetlands, but virtually absent from open water sites in the main channel of the Ross River and Ross Dam. Species diversity and abundances of indigenous fishes, however, were significantly less than for main channel sites.

There was a strong association between mass stocking of the predatory Barramundi*, Lates calcarifer*, and a significant reduction in catches between the pre-stocking (1991/92) and post-stocking (1997/98) periods, particularly of the indigenous Bony bream, *Nematolosa erebi*. Catches of the non-indigenous Mozambique mouthbrooder, *Oreochromis mossambicus,* were not affected. Piscivorous fish, including Barramundi and Mouth almighty, *Glossamia aprion,* apparently did not target non-indigenous fishes, even though the Mozambique mouthbrooder is a dominant component of the fish community in the weirs. Juvenile and subadult Mozambique mouthbrooder remained in refuge habitats inaccessible to large predators until they were large enough to join adult schools in more open water. At this stage, these fish are effectively too large (deep-bodied) and spinous to be selected as prey, even by large, gape-limited predators such as Barramundi. Experimental studies showed that piscivorous fishes, irrespective of foraging mode, were functional rather than taxonomic predators: they will eat non-indigenous fishes if available. In the absence of cover, predators chose slow, soft-finned and narrow-bodied fish as prey in preference to fast, evasive, deep-bodied fish with spines.

Non-indigenous fishes had depauperate, stochastically-determined parasite communities dominated by non-indigenous parasites introduced with the host. Non-indigenous fishes had significantly lower parasite species richness, intensity and prevalence and more aggregated parasite frequency distributions than indigenous fishes, particularly in high disturbance habitats. It is argued that these differences assist the invasion and establishment of non-indigenous fishes by reducing adverse affects of parasites on host fitness. There was a significant positive correlation between parasite acquisition and residence time for non-indigenous fishes. Parasite acquisition appears to be a very slow process and may occur over centuries – even millennia - rather than decades. This slow acquisition will also benefit non-indigenous fishes as it allows them a long period of adjustment to the new environment relatively free from parasites.

Management options for non-indigenous fishes in northern Queensland are discussed. It is argued that habitat restoration can play a key role in the long term management of non-indigenous fish species by increased exposure to predation and competition (biotic resistance) and so greatly reduce local populations of non-indigenous fishes and increase the probability of their extinction.

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CHAPTER ONE

INTRODUCTION

"Oh, - as to weeds - Mrs Gray says she allows that our weeds give up to yours. Ours are modest, woodland, retiring things, and no match for the intrusive, pretentious, self-asserting foreigners."

Excerpt from a letter to Charles Darwin
from Asa Gray, 24 November 1862
[Cambridge, Massachusetts]

This thesis examined the ecology of invasions of freshwater fishes in tropical northern Queensland. Invasions in Australia are of special interest as the continent represents one of the major biogeographic realms of the globe with its unique fauna and flora from a long period of isolation and because of the often systematic way in which people have introduced exotic species, with introductions becoming more frequent in recent years as well as an increasing awareness of the effects of such introductions on resident communities.

1.1 Historical context

Biological invasions are not a new phenomenon although it is only within the past 150 years that they have been the focus of study. From the voluminous correspondence of Charles Darwin (see Burkhardt and Smith 1985), it is clear that naturalists and scientists of the mid-19th century were fascinated by the subject of the distribution of "animal productions" and how to account for similarities in fauna between adjacent and widely separated land masses. Debates on modes of dispersal, such as icebergs, hurricanes, sea currents and land bridges, by the late 1850s, began to include human agency as an increasingly important means of new animal and plant introductions, which corresponded with the great waves of mass migrations particularly to the Americas and Australasia. There was also recognition of a lack of knowledge on the subject and of the unpredictability of the impacts of these introductions as revealed by the following excerpts from Darwin's correspondence:

i. to J.D. Hooker, 20 Jan 1859 from CD, Bromley, Kent (Burkhardt and Smith 1985, Vol. 7, p. 236):

"The only point which I *presumptuously* rather demur is about the status of the naturalised plants in Australia. I think Muller speaks of them as having spread largely beyond cultivated ground; & I can hardly believe that our European plants would occupy stations so barren that the native plants could not live there: I should require much evidence to make me believe this..."; and

ii. from Charles Darwin to H.C. Watson, January 1860 (Burkhardt and Smith 1985, Vol. 8, p. 18):

"Alph. de Condelle has shown that the species, which spread widely, tend generally to spread *very* widely, and consequently they will tend to supplant several species in several areas and thus check inordinate increase of specific forms throughout the world. Hooker has shown that in S.E. corner of Australia, where there are many invaders (i.e., dominant forms) from different quarters of the world, the endemic Australian species have been greatly reduced".

It was not until the 20th century that attempts were made to devise a conceptual framework for the study of biological invasions as the subject became of central concern to the newly emerging discipline of ecology. By combining historical syntheses with more systematic, quantitative data the aim of such a framework was to provide a more predictive model of invasions.

The terms "invasion" and "colonisation" have both been used to describe the establishment of organisms in new habitats, although it was Charles Elton (1958) in his seminal text, *The Ecology of Invasions by Animals and Plants*, who used "invasion" to emphasise the human origins of most recent introductions and the adverse impacts of many of these introductions. Elton emphasised the need to examine the past as well as the future in order to understand what is likely to happen to the "ecological balance" of the world. Elton argued that humans are now carrying on and accelerating an interchange of species that was going on some fifteen million years ago when the great continental landmasses rejoined during the Pliocene. Based on the fossil record, Elton noted that: "It is an absolute historical fact that the Pliocene invaders and the originally evolved inhabitants of the Neotropical and Nearctic Regions underwent extraordinary casualties when the two faunas had been brought together". This led to dislocations as catastrophic as modern day examples such as the invasion of the sea lamprey into the inner

Great Lakes of North America (Mills *et al.* 2000); while the time scale was different, one in millions of years and the other in decades, Elton argued that the same principles could have operated in both. As a consequence, he made the stark prediction that the eventual state of the biological world would become not more complex but simpler – and poorer: "Instead of six continental (Wallacian) realms of life, with all their minor components of mountain tops, islands and freshwaters, separated by barriers to dispersal, there will be only one world, with the remaining wild species dispersed up to the limits set by their genetic characteristics, not to the narrower limits set by mechanical barriers as well."

Elton examined recent examples of biological invasions, both natural and human-assisted, to discover if there were common principles involved which could increase prediction of individual invasion outcomes and provide for better management of the growing problem. He argued that the lack of invaders into natural systems was due to some resistance attributes that were determined by their complement of competitors, predators, parasites and diseases. Successful invasions occurred when this resistance system was in some way broken down by disturbance.

From the 1960s onward research output on biological invasions increased substantially with more emphasis on an interdisciplinary approach, including mathematical modelling (e.g., Roughgarden 1986; Hastings 1986; Williamson 1989,1996; Hengeveld 1999), prediction of outcomes and risk analyses: the reasons for the success or failure of introductions (e.g., Pimm 1989; Li *et al.* 2000) and even the anthropological, economic, legal and cultural dimensions (e.g., di Castri 1989; McNeely 1999; Shiva 1999). There was also an increasing recognition, due to the complexity and global extent of the problem, of the need for international cooperation between government and non-government environmental management agencies. There are now at least 22 global and regional multi-lateral environment and conservation treaties referencing non-indigenous species introductions (Glowka 1996; Glowka and de Klemm 1999). Besides government agencies at the national and international level, many non-government organisations (NGOs) were established with research into, and management of, invasive species as an important and even primary focus.

The Scientific Committee on Problems of the Environment (SCOPE), for example, was established in 1969 as a committee of the International Council of Scientific Unions (ICSU). It consisted of 74 founding National Academies of Science, 20 International Unions and 26 Scientific Associate bodies. Between 1971 and 1989, SCOPE addressed a wide range of concerns that involved risk and impact assessment of environmental hazards and included modelling of such hazards as anthropogenic pollutants and their effects on global biogeochemical cycles. In 1989, the SCOPE program on the Ecology of Biological Invasions focussed on invasions as a global phenomenon rather than simply a local event. The primary goal was to assess the significance of invasions into natural systems and address these questions:

- what are the factors which determine whether a species will be an invader or not;
- what are the site properties which determine whether an ecological system will be relatively prone to, or resistant to invasion; and
- how should management systems be developed based on the ecological information obtained?

The emphasis was on a functional approach: rather than viewing invaders as just taxonomic entities, it was necessary to examine the role they play in community and ecosystem processes. Elton's concepts of resistance and susceptibility to invasion were recognised by SCOPE and other researchers as an important focus for study with differing emphases on abiotic or biotic elements to explain invasion success (e.g., Simberloff 1989; Baltz and Moyle 1993; Moyle and Light 1996a,b; Naeem *et al.* 2000). Furthermore, resistance and susceptibility to invasion were considered as community-level properties, not necessarily of specific faunal assemblages (e.g., fish, birds or insects), of guilds or trophic levels: given the right set of conditions, any species or group of species could successfully establish and subsequently become "invasive". Following Elton's observations, many studies added to a growing consensus that disturbed systems tend to be more susceptible to invaders than intact systems (e.g., Brown 1989; Moyle and Light 1996a; Honnay *et al.* 1999; Ramakrishnan and Srivastava 2000), although it was recognised that disturbance might also inhibit invasions by plants and animals (e.g., Hobbs 1989; Mooney and Drake 1986; Antonio *et al.* 2001).

The 1980s witnessed the continued and alarming global decline in population numbers of thousands of plants and animal species and an increase in species extinctions. A global response was the 1992 Earth Summit in Rio de Janeiro, where 156 world governments signed the Convention on Biological Diversity (CBD) (IUCN 2000). One of the outcomes of the Convention was the recognition that next to habitat destruction, invading organisms posed the greatest threat to global biodiversity. For example, it is suggested that 20-30% of freshwater fish species are directly or indirectly threatened with extinction as a consequence of introduced species (Moyle and Leidy 1992; IUCN 2000).

Introductions have been accidental or intended to “improve” local fish faunas but, in both cases, their impacts have been adverse causing, or being implicated in, declines of indigenous fishes – what Moyle *et al.* (1986) referred to as the “Frankenstein effect.” Declines and local population and species extinctions have been attributed to introduction of predators and aggressive competitors (e.g., Crowl *et al.* 1992; Bradford *et al.* 1998; Tyler *et al.* 1998; Ogutu-Ohwayo 1990,1999), introduced parasites and pathogens (e.g., Langdon 1990; Moyle and Cech 1996; Clarkson *et al.* 1997) and hybridisation between closely related invader and resident species (e.g., Hubbs and Miller 1942; Echelle and Connor 1989). Moyle (1999) pointed out, however, the difficulties in making such assessments: in many instances the role of invaders in the decline of native fish populations worldwide may be unclear because of their close association with human-altered freshwater systems. Declines and extinctions may be due to the replacement by invaders in niches “vacated” by indigenous species as a consequence of habitat alteration, and displacement of these native species through predation or resource competition with the invaders. Either of these processes or an interaction between them may occur, and their individual effects may be difficult or impossible to separate.

At a 1996 UN conference on alien species, in Trondheim, Norway, a broad constituency of approximately 180 scientists, managers and policy-makers, as well as representatives from international organisations and NGOs (including SCOPE) engaged in the implementation of the CBD, was updated on the current research on biological invasions (Sandlund *et al.* 1996) The Conference further emphasised the human dimension of alien species across disciplines and sectors of society and the need for predictive theories that can help to set priorities for the control of such species when they become invasive and the risks of future invasions. Moyle (1996) provided an assessment of the causes and effects of invading species in freshwater and estuarine systems and of current management strategies. He noted that fresh waters are now among the most invaded ecosystems worldwide where introduced species, such as fishes, are a significant, or even a dominant component of the biota (e.g., Bruton and van As 1986; De Moor and Bruton 1988, 1996; Moyle and Williams 1990; Meng *et al.* 1994; Cohen and Carlton 1995).

Moyle argued that, although most freshwater (and estuarine) systems have been invaded, this does not mean that they are exceptionally invasible. The high frequency of successful invasions reflects the degree of alteration of aquatic ecosystems, the high frequency of introductions, many of them secondary effects of human activity (e.g., canal building and aquaculture), and high degree of successful matching between the indigenous and introduced habitats of the introduced species.

He also noted that although the ecosystem and species-diversity effects of invasives often appear small, in lakes, for example, with a high number of endemic species, the effects of introductions can appear similar to what is observed on islands with high endemism where the most devastating effects of introduced species have been observed (Strahm 1999; Clout 1999). One of the greatest ecological disasters of the 20th century was the establishment of the predatory Nile perch, *Lates niloticus*, and three herbivorous tilapiine cichlids into Lake Victoria during the 1950s and early 1960s. In the following decades, more than 300 endemic cichlid species became extinct through predation by, and resource competition with the invaders (Hughes 1986; Ogari and Dadzie 1988; Witte *et al.* 1992; Ogutu-Ohwayo 1990,1999).

With the continued growth of human populations, the globalisation of trade and climate change affecting not only local ecosystems but also the biosphere, Wittenberg and Cock (2001) argued that there was an urgent need for the development of national and global strategies to assess the full scope of the threat of invasive non-native species and deal with it effectively. This included greater resources to be provided for basic ecological research to better understand the invasion process and inform management decisions. In a world of shrinking boundaries, mechanisms for international cooperation to stop invasions at their source and to foster the sharing of lessons learned in preventing and dealing with invasions were of increasingly critical importance.

1.2 Ecological and Conceptual Framework

Essentially, a biological invasion is an event where any sort of organism successfully establishes somewhere beyond its previous range. In recent times, most invasions have resulted from human actions, either deliberate or accidental, but natural invasions also still occur, from minor changes of range to major intercontinental invasions. Ehrlich (1986) defined an “invading or colonising” species as one that easily crosses barriers, with or without the help of human beings, and rapidly establishes itself and then expands its range and numbers relatively rapidly in new habitats on the other side. Such species have been variously referred to as invaders, aliens, introductions or exotics. The term “invasive” species is applied to those which, upon entry into a new environment, spread and increase in numbers, often quite rapidly, to become a dominant and typically disruptive component in the receiving community.

There have recently been two main approaches to the study of biological invasions: analyses of case histories (Nichols *et al.* 1990; Moyle and Light 1996a) and applications of recent developments in ecological theory, especially community assembly theory (Pimm 1989; Case 1991; Townsend 1991; Lodge 1993a,b). Most invasion biology is based on a common conceptual model in which the invasion process is represented in three distinct phases: arrival, establishment and integration. Moyle and Light (1996b) provided a simple conceptual model based on community assembly theory (see Figure 1.1): that is, a community is constructed over time by a pool of potential invaders and each newly arriving species meets with what Elton (1958) called "ecological resistance" to its establishment. Moyle and Light (1996a) described this resistance as three interacting elements: abiotic (or environmental), biotic and demographic. In a freshwater system, environmental resistance includes physico-chemical factors such as water flow, chemistry, temperature and habitat structure (presence or absence of refuges); biotic resistance includes predation, competition and parasitism; and demographic resistance includes propagule pressure (numbers introduced and frequency of introduction events). Success or failure of most invasions depends on the interaction of all three elements although different models have emphasised the importance of different elements. For example, Case (1990, 1991) considered the role of biotic resistance (mainly predation and competition) to be important, while Moyle and Light considered environmental factors to be critical, at least in aquatic systems, in determining the outcome of invasions.

Williamson (1996) provided a conceptual framework that outlines the stages of an invasion (see Table 1.1), including the three main phases of invasions: arrival and establishment, spread, and "equilibrium" phase. Because of inherent differences between environments, Williamson's model needs some modification when applied to freshwater systems. For example, in isolated waterbodies, secondary human translocations may be far more important than natural dispersal to increase the range of an introduced species. Williamson emphasised the importance of biotic resistance, where invaders interact with natives through vertical (e.g., predation) and horizontal (e.g., competition) processes, which may result in population declines and extinctions of indigenous species and even ultimately to ecosystem restructuring. However, he also recognised that several abiotic and demographic factors need to considered in assessing or "predicting" the success of an invader: the intrinsic rate of natural increase (r), abundance in native habitat, taxonomic isolation, climatic and habitat matching and presence of vacant niches in the receiving environment.

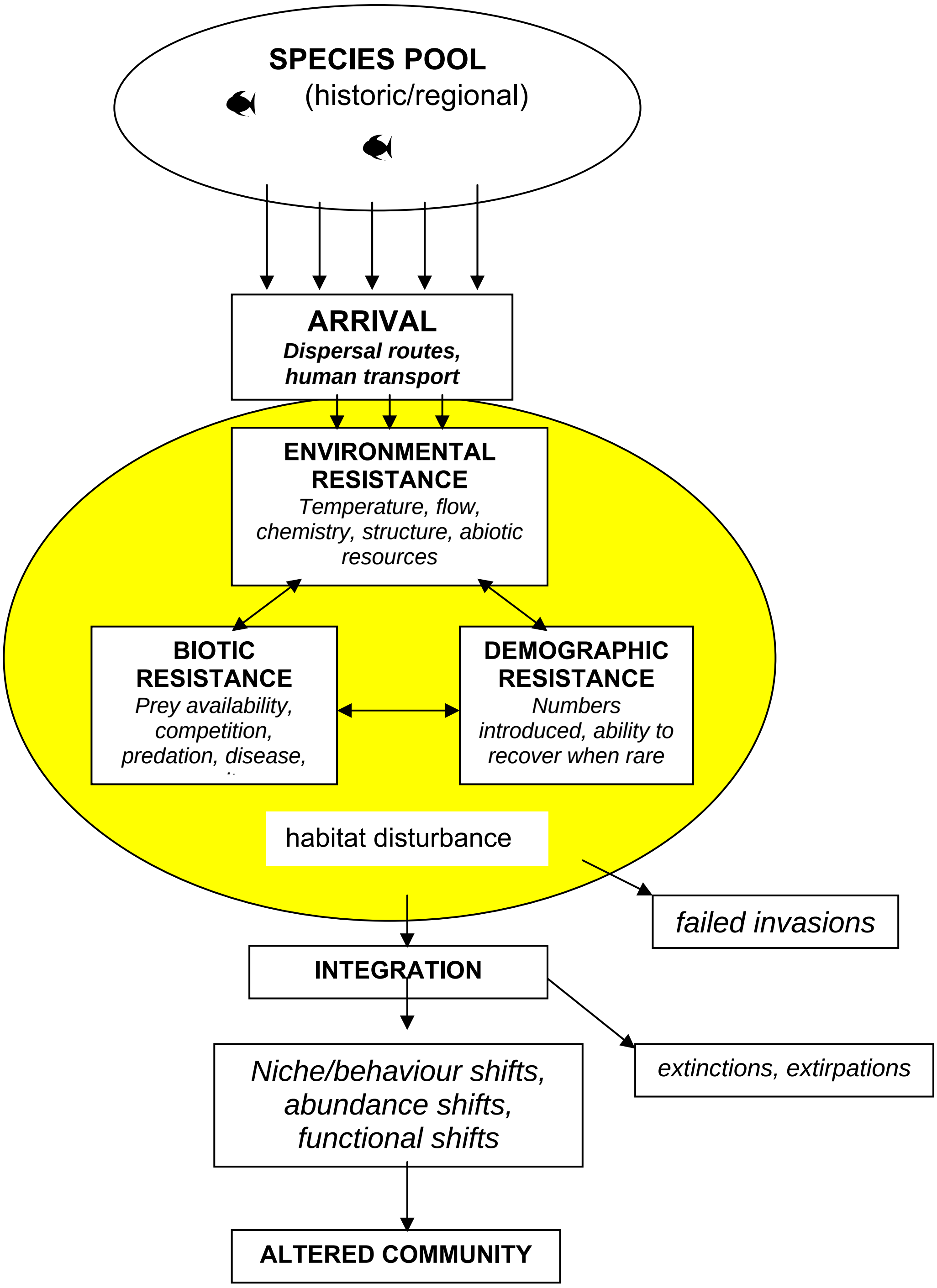


Figure 1.1 A conceptual model of invasions based on Community Assembly Theory (modified after Moyle and Light 1996a)

Table 1.1 A conceptual framework for biological invasions (from Williamson 1996)

A. Arrival and establishment	
0	Most arrivals at present are from human importations, but natural arrivals are also of interest
1	Most invasions fail, only a limited number of taxa succeed (tens rule)
2	Invasion (or propagule) pressure is an important variable. So invasions are often to accessible habitats by transportable species
3	All communities are invasible, perhaps some more than others
4	The a priori obvious is often irrelevant to invasion success. Among factors to consider: r (intrinsic rate of natural increase), abundance in native habitat, taxonomic isolation, climatic and habitat matching, vacant niche.
B. Spread	
5	Spread can be at any speed in any direction, in analysed cases usually as predicted by estimates of r and D (diffusion coefficient) or faster
C. Equilibrium and effects	
6	Most invaders have minor consequences (tens rule)
7	Major consequences have as • effects: depressed populations to individual extinctions to ecosystem restructuring • mechanisms: enemies (vertical food chain processes), competition, amensalism, swamping (horizontal food chain processes
8	Genetic factors may determine whether a species can invade; genetic factors affect events at the initial invasion; evolution may occur after invasion
D. Implications	
9	Invasions are informative about the structure of communities and the strength of interactions, and vice-versa
10	Invasion studies are relevant to considering the risks of introducing new species or genotypes, the release of genetically engineered organisms and the success and consequences of biological control.

Ehrlich (1986) and Moyle (1986a) had previously suggested a number of similar *pre-* and *post-invasion* attributes that one could expect to characterise successful invaders:

pre-invasion attributes

a) abundant within a wide original range
b) broad ecological tolerances

c) have reproductive strategies which confer on them an unusual degree of fitness and much genetic variability
d) aggressive, so can eliminate native species through a combination of predation and competition
e) well-established association with humans
f) polyphagous
g) larger than most relatives
h) able to disperse and colonise new areas rapidly

post-invasion attributes

i) they are preadapted to local or regional conditions
j) they are ecologically or behaviourally distinct from native fishes, so the latter do not interact with them or are unable to deal with a new style of predation or competition.

Most established invaders are not pests and are not successful in spreading from the original point(s) of arrival into a wide range of habitats and do not form large populations. The reasons for this are not well understood and predictions of the outcome of new invasions remain, as yet, limited and unreliable. Lodge (1993b) suggested that ecosystems susceptible to invasion were climatically similar to the original habitat of the invader, were relatively simple with many vacant niches, had low diversity and an absence of predators or grazing species. Moyle and Light (1996a) developed potential "rules" for predicting the effects of invasions in fresh waters. They suggested that virtually any species can invade and that any ecosystem can be invaded given the right circumstances. They considered that the most dramatic invasions usually occur when the invader is a piscivore or planktivore and when the invaded ecosystem has relatively low natural diversity. Furthermore, the effects of an invasion are more likely to be reduced by environmental resistance than by biotic resistance of the established community. Several studies (e.g., Fox and Fox 1986; Lodge 1993a,b; Moyle and Light 1996a) have suggested that habitat disturbance may have a significant role in successful invasion. Fox and Fox (1986) argued that many exotic species are more successful than indigenous species at exploiting changed resource conditions caused by anthropogenic disturbance of habitat, particularly where the invaders have attributes of trophic plasticity (omnivory) and wide physiological tolerances (Ehrlich 1986; Moyle 1986; Lodge 1993a). Moyle and Light (1996a) also noted that changed habitat may benefit invading species by reduction in biotic diversity (reducing numbers of potential native predators or resource competitors).

Dove (1999) observed that patterns of resource alteration are usually one of several specific interactions between exotic, non-indigenous species and indigenous species, including predation, competition and introduction of exotic pathogens. A number of studies have demonstrated that exotic plants and animals, including fish, lose a significant proportion of their co-evolved parasites when introduced to new habitat (Dogiel 1948; Petrushevski 1961; Dove 1999; Torchin *et al.* 2003; Mitchell and Power 2003; Clay 2003). Dove (1999) argued that such a reduction advantages the invader directly, by reducing the negative impact of the parasites on host fitness, and indirectly, where indigenous hosts are infected by parasites or pathogens imported with the invader (Langdon 1990; Bauer 1991). Torchin *et al.* (2003) and Mitchell and Power (2003) even suggest that the success of some invasive species may be strongly associated with their loss of parasites and consequential increase in vigour when entering new environments. Dove (1999) suggested that habitat disturbance may also affect the host-parasite interaction and further advantage the host invader where conditions adversely affect the survivorship or transmission of the parasite rather than the establishment and long-term persistence of the host. He concluded that invading species should be regarded as invading communities since the host is likely to bring with it at least some of its co-evolved parasites. Dove argued that regarding non-indigenous fish species as host-parasite systems complicates considerations of their effects on indigenous fishes, but may provide a more realistic conceptual model of species invasions than models that exclude the role of parasites.

While improved prediction of establishment success and impacts of non-indigenous species is desirable, some researchers have added a cautionary note. Williamson (1996) pointed out that ecosystems are complex and biological processes can be exceedingly variable between species, so that the *a priori* obvious is often irrelevant to invasion success. Moyle and Light (1996a,b) noted that stochasticity appears to be a significant factor in invasion success: invading species and invaded ecosystems have idiosyncrasies that often defy accurate prediction.

Wittenberg and Cock (2001) stated that a comprehensive knowledge base is crucial in the development of national strategies to deal with introduced organisms, especially those that become invasive. Information required includes well documented accounts of species distribution, past spread and potential future dispersal; their impacts; “synergistic” effects and other interactions between non-indigenous and indigenous species; and an understanding of major pathways of introduction of non-indigenous species. Furthermore, they argued that problems caused by invasive species are increasing dramatically. These impacts are likely to change and probably intensify over time in response to anthropogenic environmental changes, including land and water use practices and global climate change due to greenhouse warming, and that monitoring and research needs to be continuous and well-supported. In Australia there is still remarkably little known about the basic ecology of indigenous freshwater organisms as the national focus has been largely terrestrial or marine. There have been a few ecological studies of non-indigenous fishes in Australia (e.g., Jackson and Williams 1980; Fletcher *et al.* 1985; Arthington and Milton 1986, Webb 1994; Arthington and Marshall 1999) and reviews on their impacts (e.g., Fletcher 1986; Morison 1989; Arthington 1989), though knowledge has relied to a large extent on accounts, where available, from the indigenous ranges of these fishes (e.g., Lowe-McConnell 1982; Grether *et al.* 2001; Heibo and Vollestad 2002) or their introduced ranges elsewhere (e.g., Man and Hodgkiss 1977a,b; De Silva 1985; Garcia-Berthou 2001; Zambrano *et al.* 2001). Attention has also recently focussed on the importance of non-indigenous fishes in Australian waters as vectors of disease agents, such as viruses and bacteria (Langdon 1990; Whittington *et al.* 1996; Reddacliff *et al.* 1996) and macroparasites (Dove *et al.* 1997; Dove 1998,1999), and on the development of models to assess the risk of establishment and invasiveness of non-indigenous fishes in Australia (Arthington *et al.* 1999).

1.3 Aims and Objectives

Using case studies of non-indigenous species in particular waterways, and experimental studies, the aims were to determine likely reasons for the successful establishment of non-indigenous fishes in northern Queensland. Essentially, this study aimed to test current ideas regarding the importance of biotic resistance, habitat, and parasitism in establishment of non-indigenous species rather than an investigation of their invasiveness, that is, their adverse impacts on indigenous aquatic communities. Tropical northern Queensland is ideally situated to test these ideas because:

- the majority of non-indigenous fish species imported into Australia during recent decades for the aquarium trade are tropical species and therefore have a high probability of successful establishment in the region;
- the majority of tropical species released into open waters in Australia occur in the region;
- many waterways where these fish occur, especially in urban and agricultural areas, have experienced varying levels of human disturbance, although there remain, for comparison, a number of catchments and sections of waterways still relatively undisturbed; and

- the history of introductions in the region is reasonably well-documented.

To achieve these aims, the main objectives were to:

1. determine the number, distribution, mode and rate of dispersal of non-indigenous fishes in northern Queensland;
2. gauge the success of establishment of breeding populations by non-indigenous fishes using historical data and indices of propagule pressure and habitat matching;
3. test the importance of predation pressure on the occurrence and population structure of non-indigenous fish populations after mass stocking of a predator in an impounded river system, including a comparison of pre-stocking and post-stocking catch data, a description of the diets of predatory fishes, and use of refuges as an anti-predator strategy by non-indigenous fishes; and
4. describe "post-invasion" changes in parasite community structure of non-indigenous freshwater fishes in northern Queensland, including changes to parasite species richness, occurrence of non-indigenous parasites and their aggregation patterns;
5. estimate the rate of parasite acquisition by non-indigenous fishes in the tropics and the effects of habitat disturbance on parasite acquisition; and
6. discuss the possible significance of (4) and (5) to host fitness and, therefore, establishment success and persistence of non-indigenous fishes.

This thesis addresses these aims and objectives in the following chapters:

Chapter 2 describes the study region and study sites;

Chapter 3 examines the distribution and success of non-indigenous fishes in northern Queensland;

Chapter 4 examines the distribution and abundance of non-indigenous fishes in the Ross River catchment, in relation to habitat and presence of predators;

Chapter 5 describes experimental studies of predator-prey interactions between indigenous piscivorous fish and non-indigenous fish species, including the effectiveness of refuges from predation;

Chapter 6 compares the parasite faunas of indigenous and non-indigenous fishes in northern Queensland and examines the effects of habitat disturbances on the host-parasite relationship; and

Chapter 7 provides a synthesis of the results and their implication for management of non-indigenous fishes.

Appendices A – E provide supplementary information.

CHAPTER TWO

GENERAL DESCRIPTION OF STUDY AREAS AND SITES

"The country tho in general well enough clothd, appeared in some places bare; it resembled in my imagination the back of a lean cow.."

Sir Joseph Banks (*Journal entry: description of Queensland coast*)
25 April 1770

2.1 Northern Queensland – general description

Northern Queensland can be defined biogeographically by factors such as climate (including rainfall), geology, drainage patterns and vegetational and faunal communities. Figure 2.1 shows the five major freshwater drainage divisions of Queensland based on the Australian Water Resources Council (1988) classification system according to the orientation and geographic areas covered by their drainage patterns and the main climatic zones (Zeller 1998). The Northern Region includes the two major drainage divisions: the North East Coast and Carpentaria from the tip of Cape York Peninsula southward to include the Burdekin River catchment. The major landform defining these divisions is the Great Dividing Range. Coastal streams draining its eastern side empty into the Pacific Ocean while those draining westward north of the Gregory Range (inland from Townsville) and those draining the eastern and northern sides of the Western Plateau (Barkly Tableland) flow into the Gulf of Carpentaria.

Climatically, northern Queensland can be broadly divided into three regions based on rainfall: the Dry tropics, the Wet-Dry Tropics and the Wet Tropics. The Dry Tropics is where mean annual rainfall is less than 500 mm in the central region inland of Townsville, in the Selwyn Range and eastern Barkly Tableland. This region has unreliable rainfall and is prone to drought, although it may experience flooding depending on the position of the monsoonal trough during the summer months. The Wet-Dry Tropics is where annual rainfall is 600-1600 mm (on average 1,000 mm) on southern Cape York and coastal areas south of Townsville to Mackay. The Wet Tropics is where annual rainfall is in the range of 1600-4000 mm, on the eastern coastal plain, tablelands and mountains of the Great Dividing Range from Cooktown to Townsville.

2.1.1 Study area: Townsville region

Since its establishment as a port in 1864, Townsville has undergone rapid growth, particularly following the discovery of gold in Charters Towers in 1872 and in the

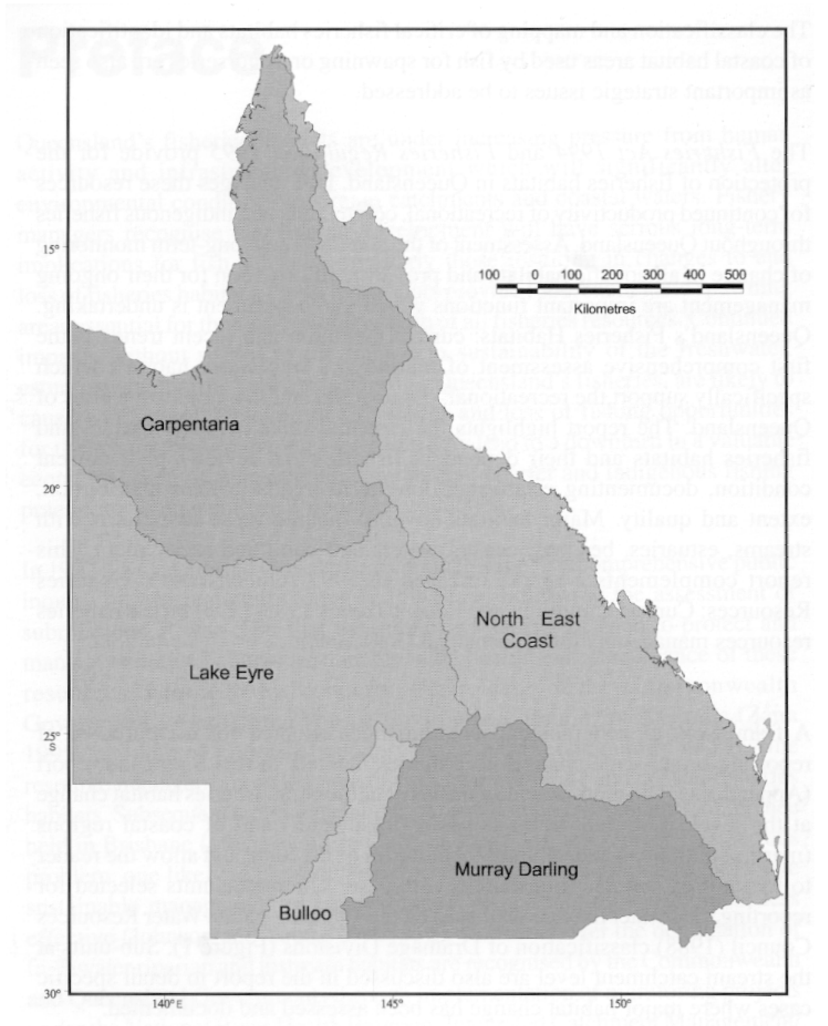


Figure 2.1 Freshwater drainage divisions of Queensland
(© The State of Queensland, Department of Primary Industries, 1998, used by permission) (Zeller 1998)

period after the Second World War. In 1976, the Townsville district population was approximately 80,000 increasing to 125,000 by 1997, and with most of the recent suburban expansion taking place in Thuringowa Shire, the latter gaining City status in 1986 (Humphreys 1978; Hornby 1993; Australian Bureau of Statistics 2000). The twin cities are now the third largest conurbation in Queensland (following Brisbane and the Gold Coast) and the largest urban centre in tropical Australia. As a consequence of this development, the local waterways, in particular the Ross River and its distributary the Ross Creek, have become highly modified environments. The twin cities of Townsville and Thuringowa are located on a narrow coastal plain (S19˚ 16', E 147˚ 49') (Figure 2.2). The plain is approximately 30 km wide to the south of Townsville, defined by the Mount Elliot massif, and narrows to approximately 5 km wide to the north at Saltwater Creek. The plain rises gently to the foothills of the coastal escarpment, then rises abruptly to 1000m in the north (Paluma Range) but to only 650 m further south (Hervey Range). The rivers to the north are short and of low order, flowing directly to the sea, while the relatively larger systems, the Black, Bohle and Ross River, occur where the plain widens to the south. Like the smaller rivers to the north, the headwaters of these systems are essentially restricted to the foothills of the coastal escarpment.

The coastal plain consists largely of strongly weathered clay deposits (kaolinite) sloping gently seaward and dominated by poor solodic soils (Hopley and Murtha 1975). From the plain rise residuals of the original, eroded bedrock, such as Many Peaks Range and Castle Hill. Fluvial deposits of sands and gravels occur along all the watercourses and are present as narrow floodplains or levees. Beach ridges are present along the entire coastline and consist of solodic soils, mangrove muds, beach rock and gravel, with salt water couch meadows and salt pans found behind the beach ridges. Mangroves line all estuaries and areas of mangroves are extensive in the lower Bohle River and creeks to the south, draining into Cleveland Bay and Bowling Green Bay (Haughton River and Barrattas Creeks).

2.1.2 Cairns Region

Cairns is located on the western shores of Trinity Inlet on the narrow (6 km), flat coastal plain (S16˚ 52', E145˚ 45') (Figure 2.2). The city was established in 1876 following the discovery of gold on the Hodgkinson River in the Mulligan Range. Subsequent growth of the township was due largely to the establishment of the sugar industry in the 1880s. The population of Cairns in 1947 had reached 16,000 increasing to 35,000 by 1975. The City became a regional service centre for Far North Queensland, although its recent development has been largely due to the tourism industry.

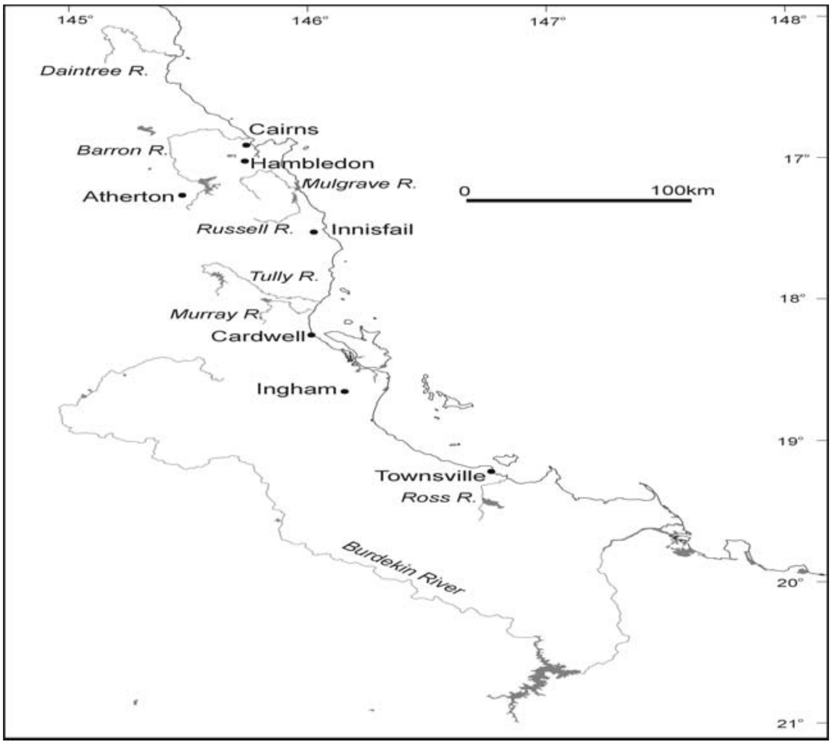


Figure 2.2 General location map of northern Queensland

The numbers of international visitors to Cairns increased from 29,000 in 1975 to 320,000 per year by 1988, representing 14% of the total number of visitors entering Australia, and to an average of 437,000 for the period 1997-1999 (P. Jankowski, Cairns Port Authority, *pers. comm.*). Consequently, the population increased rapidly during this time to 81,000 by 1989, a 359% increase over the preceding 40 years. In 1997, the population reached 110,000 (Australian Bureau of Statistics 2000).

The Cairns region is dominated by sedimentary rocks of the Hodgkinson formation of the tablelands to the west, with granitic protrusions forming a series of mountainous ranges encircling Cairns. These ranges are cut by gorges with short, fast flowing streams, such as Freshwater Creek and the Barron River. The headwaters of Freshwater Creek have been dammed to form Lake Morris, which is the water supply for Cairns. The creek joins the Barron River to the north of Cairns, downstream from the Barron Falls and Gorge. Below the Gorge, the river slows considerably before it enters Trinity Bay through a large delta area. Extensive sedimentation from the Barron River has allowed the development of the tidal wetlands along the Esplanade of the City.

The coastal area from Cairns south to about Cardwell is within the southern section of the Queensland Wet Tropics biogeographic region. It is delineated in the west by the foothills of the Great Dividing Range. The area consists of relatively flat depositional plains and extensive beach ridge systems. The majority of these plains have been cleared for agricultural use including sugar cane, bananas, small crops, cattle grazing and forestry, although the latter has declined in importance due to the inclusion of large remaining tracts of forest within the World Heritage Estate. Inland on the foothills there is closed vine forest or rainforest with open and closed sclerophyll (eucalypt) forest and woodland on steep ridges. This vegetation, also on the extensive coastal beach ridges and associated low-lying freshwater swamps and lagoons, is dominated by Swamp mahogany, *Eucalyptus robusta*, and species of *Melaleuca*, *Casuarina* and *Acacia* (Cannon *et al.* 1992). South from Cardwell the coastal plain is incised by rivers and streams that run from the Dividing Range to the sea. North of Cardwell, there are more active alluvial systems with historically more variable stream courses. For example, the Murray River was previously a distributary of the Tully River and both rivers share a common alluvial plain. The Russell, Mulgrave and lower Barron River systems have also frequently changed course due to high rainfall causing flooding and sediment deposition (Stephenson and Willmott 1989).

The coastal plain immediately north of Cairns between Yorkey's Point and Buchan Point narrows to 1-2 km wide and is intersected by a number of small streams which enter a series of small melaleuca swamps behind dune ridges. Downstream are mangrove estuaries, which are intermittently blocked by sandbars, forming brackish lagoons. The coastal plain to the north of Cairns is interrupted at intervals by steep mountain ranges, such as the McAllister Range and Dagmar Range. The streams flowing off these ranges are short, steep and intermittent, except where the plain widens, e.g., Hartley's Creek, Tin Creek, the Mossman River and Saltwater Creek systems. These represent the only permanent, flowing freshwater between Buchan Point and the Daintree River, a distance of 50 km. The narrow coastal plain continues north of the Daintree River to Cape Tribulation and is intersected by a series of short, fast-flowing, permanent streams. This area contains some of the last extensive remnants of well-developed lowland rainforest in northern Queensland, although it has experienced increasing pressure from human activity, through tourism and rural sub-division.

2.1.3 Atherton Tablelands

The Atherton Tablelands (Figure 2.2) occupies an area of about 160,000 ha to the west of the coastal plain between Cairns and Innisfail. It is a plateau 700-900 m above sea level and consists of gently sloping rises in the north to steeply sloping low hills (30-90 m) and higher hills (90-300 m) in areas to the south. The geology is dominated by basaltic and granitic soils and volcanic cinder cones and crater lakes (e.g., Lake Eacham and Lake Barrine) that are prominent landscape features. Malcolm *et al.* (1999) described the climate as humid sub-tropical modified by the elevation. Rainfall is highly variable although there is a gradient from the higher altitude areas to the southeast with levels in excess of 3500 mm annually around Millaa Millaa, Topaz and Bellenden Ker to around 1100 mm on the gentler sloping rises to the north and west around Herberton, Tolga and Mareeba. Mean maximum temperatures are between 20 and 30ºC and minima between 9 and 20ºC, with temperatures as low as –4ºC recorded in the Herberton District.

The original vegetation was dominated by rainforest with sclerophyll woodland in drier areas (Tracey and Webb 1975; Tracey 1982). The boundary between the two follows the 1400 mm-rainfall isohyet and occurs around Tolga in the north, Kaban in the south and the Herberton Range area to the west (Kershaw 1974; Isbell *et al.* 1976). Between the two are transitional areas or ecotones consisting of mixed communities of closed vine forest and sclerophyll species. Most of the forest on lower sloping areas was cleared for the timber industry and agriculture including dairy and beef cattle farms with the start of European settlement from about 1870 onwards. According to Winter *et al.* (1987), by 1980 more than 76,000 ha of forest had been removed and Laurance (1991) reported the remainder as occurring in scattered fragments ranging in size from 1 to 600 ha. There are two major catchments: the Barron River and the

Johnstone River catchments. The Barron River drains the area north of Malanda while the North and South Johnstone Rivers drain the area to the south of Malanda. All rivers have had their courses altered in the past by volcanic activity. In the 1960s the hydrology of the Upper Baron River was altered by the construction of the Tinaroo Dam that supplies water to the Mareeba/Dimbulah Irrigation area and the Atherton township. Many small streams throughout the Tablelands have also been highly modified by construction of weirs and loss of riparian vegetation.

2.2 Climate of northern Queensland

The coastal regions of tropical northern Queensland experience a marine-modified monsoonal climate of hot, humid summers with high rainfall and relatively dry, mild winters. According to Hopley and Murtha (1975), the winter pattern is dominated by the semi-permanent anti-cyclonic high-pressure system of the South Pacific and its southeasterly trade winds, which are normally associated with fine, dry weather. The summer pattern is dominated by the establishment and position of the Monsoonal Trough or Inter-Tropical Convergence Zone that produces heavy cyclonic rainfall. The mean southerly extension of the Trough occurs in the vicinity of Ingham, so that areas to the north tend to receive more reliable summer rains that may be reinforced by orographic falls associated with the trade winds. When the trough establishes further south, Townsville may experience heavy rainfall and increased risk from tropical cyclones.

Long-term records from the Bureau of Meteorology show that average air temperature patterns for the northern region are similar, though slightly higher for Cairns. For Townsville, the average maximum monthly temperature (January) and minimum temperature (July) are 31.3˚C and 13.6˚C respectively, while for Cairns the average maximum and minimum values are 31.4˚C and 17.1˚C respectively. Maximum daily summer temperatures may be very high, particularly in Townsville, when there is less cloud cover compared with Cairns and the sun is near to, or actually, overhead. The maximum recorded temperature for Cairns is 40.5˚C (December) while the maximum for Townsville is 44.3˚C (January). Townsville has, on average, fewer cloudy days and more clear days per month compared with Cairns especially for the period from December to March (21.9 clear and 49.8 cloudy days for Townsville compared with 15.6 clear and 44.2 cloudy days for Cairns). As the wet season develops, cloudiness increases, though solar radiation declines from February onwards as the relative position of the sun moves from overhead and reaches a minimum in June at the winter solstice. The actual number of sunshine hours, however, increases on average from a minimum in February in both Townsville and Cairns to a maximum of 9 hours in August in Townsville and 8.8 hours in October in Cairns, coinciding with average monthly cloudy day minima for the two locations.

The winter dry seasonal pattern may be modified by occasional weakening of the high-pressure system that brings light orographic rainfall along the coast particularly between Ingham and Cairns. In July 1991, for example, there were two five-day cold snaps where minimum temperatures dropped to between 5.1˚C and 7.7˚C (av. 6.4˚C) and between 8.5˚C and 9.3˚C (av. 8.6˚C) respectively. The overall temperature range for the month was 5.1˚C to 18.5˚C (av. 12.4˚C). The lowest minimum winter temperature recorded for Townsville was 1.1˚C (August) while in Cairns the lowest minimum temperature recorded was 6.2˚C (June).

Intra-seasonal and inter-seasonal changes in the Australian climate are strongly influenced by the global climate system referred to as El Nino – Southern Oscillation (ENSO). This complex system, due to swings in the mean air pressure difference across the Pacific, produces periods of abnormally low rainfall across eastern and tropical northern Australia approximately every 3 to 8 years. In the tropics when the SO index (SOI) is strongly negative (El Nino phase), the monsoonal season is usually late with relatively low rainfall, followed by a period of very heavy rains when the SOI enters a strong positive phase (La Nina) (Bureau of Meteorology 2000). This occurred in the summer of 1997 after a period of about 5 years of unusually low rainfall following the previous ENSO event in the period 1989-91. The annual rainfall for Townsville for the 1992-96 period was between 469 and 710 mm. For the 1997-2000 period, the annual rainfall was between 1076 and 2399 mm.

2.3 Description of main study area: the Ross River catchment

2.3.1 The Ross River

The Ross River is a highly modified system consisting of headwaters, a dam, three weirs and an estuarine section. The three weirs (Aplin Weir, Gleeson's Weir and Black Weir) are sections of the river impounded by the construction of barrages across the Ross River in 1912 and 1934, to provide a water supply for Townsville. They were superseded by the construction in 1975 of Stage 1 of the Ross River Dam at the confluence of the river with Five Head Creek (Figures 2.3 to 2.5). These barriers (the barrages and the dam wall) have dramatically altered the hydrology of Ross River, restricting the tidal influence, increasing water depths and altering stream flows, flood discharge rates, sediment deposition patterns and general water quality. The construction of the weir barrages has effectively created a series of long, relatively narrow lacustrine environments that, in addition to other human activities, has had significant impacts on native aquatic fauna and flora. The first barrage divides the Ross River into distinct salt and freshwater regimes. The first weir, Aplin Weir, is 5275 m in length, with an average width of 212 m and surface area of 83 ha, and a capacity of 1773 Ml when full. The second weir, Gleeson's Weir, is 1250 m in length, with an average width of 150 m and surface area of 14 ha, and a capacity of 588 Ml. The third weir, Black Weir, is 6450 m in length and ends at the Ross Dam spillway. The weir has an average width in its lower section of 212 m, narrowing to about 100 to 50 m in its upper reaches, with a surface area of 83 ha and a capacity of 1640 Ml.

The riparian vegetation has been highly modified with large-scale clearance for housing and parkland development and loss from flood erosion, with subsequent invasion by exotic species, including Paragrass, *Urochloa mutica*. There has been, however, some restoration of indigenous vegetation at a few sites by local Council and community groups. The riverbed and banks have been modified by sand extraction at several locations. Small-scale operations began around the beginning of the last century, with increased activity during the war periods and construction of the barrages. In the 1980s large-scale commercial sand extraction began using draglines and excavators. Aplin weir was drained for six months for construction of a pipeline in 1980. Extraction continued for another three to four years in the weir then operations shifted to a site immediately above the Black Weir in the late 1980s. Some localised dredging was also done upstream from this site over a 1km reach near the Apex sampling site from 1995 onwards to deepen the river and remove dense mats of floating Water hyacinth, *Eichornia crassipes*.

Other large-scale commercial sand extraction began in 1994 in the upper Ross River above the Ross Dam. Some localised dredging in the Aplin Weir occurred in 1998 with construction of a new section of the Charles Barton Bridge and between 2000 and 2002 to deepen the river for the Townsville Rowing Club opposite the club's headquarters about 300 m upstream from the bridge (I. Boyce, Dept. Natural Resources and Mining, Townsville, *pers. comm.*). There are a number of small streams that flow into the Ross River, most of which are temporary drainage lines, either drying completely or remaining as a series of small pools in the dry season. These small streams can provide important breeding habitat in the wet season for a number of fish species including the Black catfish, *Neosilurus ater*, and Hyrtl's tandan, *Neosilurus hyrtlii* (Orr and Milward 1984) (Figure 2.6). These streams, however, have been highly modified, in some cases serving primarily as stormwater drains with removal of riparian vegetation and with concreted streambeds to accelerate runoff. Some of these streams are overgrown with aquatic vegetation, such as Paragrass, and their entrances blocked by this invasive species along with extensive mats of other exotic plants, notably Water hyacinth (see Chapter 4).

2.3.2 Fish community

Prior to this study, at least 23 species of indigenous freshwater fish and six species of non-indigenous fish were recorded in the freshwater reaches of the Ross River catchment (Webb 1994; Webb unpubl. data) (see Appendix A, Table A1). No descriptions are available of the fish community that was present before construction of the Ross Dam and barrages, although anecdotal evidence suggests that major changes have occurred. These changes include the disappearance in the freshwater reaches of catadromous species such as Barramundi, *Lates calcarifer*, Jungle perch, *Kuhlia rupestris*, and Tarpon, *Megalops cyprinoides*, because of the physical barrier to upstream migration created by the barrages. During the past 25 years, there have been several introductions of indigenous and non-indigenous fish species into the river. Introductions of indigenous fish, both authorised and unauthorised, have occurred for recreational fishing. These include the Sooty grunter, *Hephaestus fuliginosus,* Sleepy cod, *Oxyeleotris lineolatus*, Golden perch, *Macquaria ambigua*, and Barramundi (Gillies 1978; Webb 1994). Of these, only the Sleepy cod has established a successful breeding population in the weirs and dam. The largest stocking has been of Barramundi with approximately 51,000 fingerlings stocked between 1992 and 2000 by the Queensland Fisheries Service (M. Pearce, Northern Fisheries Centre, Cairns, *pers. comm.*). The six species of non-indigenous fish that have been recorded from the Ross River catchment are three poeciliids and three cichlids (Webb 1994). Of these, the tilapiine cichlid, the Mozambique mouthbrooder, *Oreochromis mossambicus*, is a dominant component in number and biomass of the fish community in this catchment and others within the Townsville-Thuringowa region (see Chapter 4).

Instream modifications (e.g., desnagging) has most probably lead to the decline and disappearance of indigenous species such as the Snakehead gudgeon, *Ophieleotris aporos*, due to loss of spawning and refuge sites. The introduction and rapid spread in the 1970s of exotic water plants, notably Salvinia, *Salvinia molesta* and the Water hyacinth, and the subsequent herbicide-spraying program, probably affected many fish species, including surface feeding insectivorous fish such as the Archer fish, *Toxotes chatareus*. This species was common in the weirs, but disappeared within a few years of the introduction of Salvinia. The species has subsequently reappeared in the weirs (now virtually free of Salvinia, though not Water hyacinth) in the 1990s, most probably by natural downstream migration from a surviving population in the Ross River Dam.

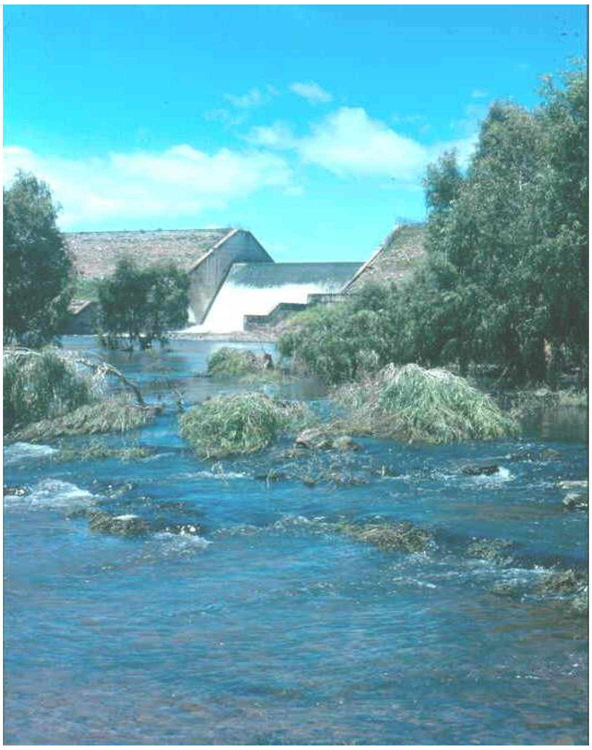


Figure 2.3 Ross River Dam spillway during the Wet Season

Figure 2.4 Aplin Weir barrage, Ross River (estuarine reach downstream)

Figure 2.5 Aplin Weir, Ross River, view upstream from barrage (Note: entrance to side channel on right blocked by Lotus lilies: Bush Garden site, Chapter4)

Figure 2.6 Campus Creek at the start of Wet Season rains, December 2002 (Note: fishway in R.H. barrel of culvert)

2.3.3 Other Ross River vertebrate fauna

During the study, the following piscivorous vertebrates were observed: Krefft's river turtle, *Emydura krefftii*, Johnstone River crocodile, *Crocodylus johnstoni,* Little File snake, *Acrochordus granulatus*; Water rat, *Hydromys chrysogaster;* and several piscivorous waterbirds including the Large egret, *Egretta alba*, Darter, *Anhinga melanogaster*, Pied cormorant, *Phalacocorax varius*, Little pied cormorant, *Phalacrocorax melanoleucos,* Little black cormorant, *P. sulcirostris,* Crested grebe, *Podiceps cristatus,* Australian pelican, *Pelecanus conspicillatus*, Brahminy kite, *Haliastur indus*, White-breasted sea eagle, *Haliaeetus leucogaster* and Osprey, *Pandion haliaetus.* Fleay (1978) recorded 41 species of waterbirds on the Ross Dam, including at least 23 piscivorous species.

2.3.4 Riparian and instream vegetation

As a consequence of continued housing and parkland development there remain only small, narrow patches of relatively intact riparian vegetation along the Ross River. Along much of the open, exposed riverbank and banks of the streams that flow into the main channel, the introduced Paragrass has invaded and formed dense mats in the shallow margins, or in some cases, along with Water hyacinth, completely blocked the watercourse. Further clearance and modification of the native riparian vegetation is occurring, although the local council and community groups are carrying out some rehabilitation work. During this study and a previous survey by Webb (1994) at least 24 species of aquatic plants were recorded from the Ross River above the Aplin Weir barrage (see Appendix A, Table A2). Although no historical data are available for the aquatic vegetation prior to construction of the dam and weirs, significant changes have occurred due to the altered hydrological conditions. A major change was the introduction of non-indigenous plants including several floating plants such as Salvinia*,* Water lettuce, *Pistia stratiotes,* and Water hyacinth, and more recently, the submerged aquatic plant, the Fanwort, *Cabomba caroliniana*, that has become widespread in the weirs. The floating plants, originally introduced as ornamental plants for outdoor ponds, probably entered the river after flooding, while the presence of Fanwort may be due to disposal of the contents of home aquaria. The floating plants, in particular, flourished in this high nutrient lacustrine environment and in the late 1970's and early 1980's reached major infestation levels. Management by the local City Council has involved spraying with herbicides as well as biological control with the introduction of herbivorous weevils. Successful control of Salvinia by the weevil, *Cyrtobagus singularis*, was achieved following its introduction in the mid-70's but similar control has not yet been possible for the other exotic plants and regular spraying with herbicides continues. The local City Council has recently started small-scale harvesting of vegetation in some of the weirs as a short-term management option. Translocated indigenous plants, e.g., the Lotus, *Nelumbo nucifera*, and local species, e.g., Hornwort, *Ceratophyllum demersum,* and Water thyme, *Hydrilla verticillata,* have proliferated in the weirs.

2.3.5 Water quality

The physico-chemical characteristics of a waterbody are a complex interaction of a wide range of factors including the geology of underlying and upstream substrate, climate and biota as well as activities related to human settlement within the catchment. Summaries of monthly water quality data (1994-2000) for the Ross Dam, Black Weir and Aplin Weir are presented in Appendix A, Figures A2 to A24.

MANOVA analyses of water quality parameters for the period 1994-2000 (Appendix A, Tables A3 to A5) show that the three water bodies have distinct and consistent differences in water quality characteristics particularly in relation to pH (alkalinity, bicarbonate concentrations) and salinity or conductivity (sodium and chloride, and other dissolved ions such as calcium and magnesium), and nutrients (total nitrogen and phosphorus). The relatively high conductivity is related to the underlying sodic soils of the Ross catchment with minerals leaching into the water or being carried downstream along with nutrients (nitrogen and phosphorus) in sediments during periods of heavy rain. Between 1992 and 1997, there was no overflow from the dam into the weirs due to below average wet season rainfall. Aplin weir had significantly higher conductivity, pH and trace element concentrations (e.g., calcium and magnesium) compared with Black Weir and Ross Dam. During this period the Ross Dam and Black Weir immediately downstream of the Dam became increasingly similar in water chemistry.

The change between these two periods corresponded with an increase in summer rains (see Appendix A, Figure A1) associated with the interseasonal periodicity of the El Nino Southern Oscillation (ENSO) phenomenon and water overflowing from the dam into the weirs. There were virtually no significant differences in water quality parameters between weirs for the period 1997 to 2000, although differences, particularly in conductivity, between the Ross Dam and downstream weirs began to re-establish during this period.

The monthly turbidity data for Aplin weir during this period revealed no significant elevation in suspended sediment levels which could be attributed to localised dredging and bridge work (Appendix A, Figure A10). From water quality data analysis for the period 1994 to 2000, mean annual turbidity levels were higher in the Black Weir and Ross Dam compared with Aplin Weir located further downstream and were significant in four out of seven years (Appendix A, Figure

A24). These differences are probably associated with the more extensive dredging activity in Black Weir and the Upper Ross compared with Aplin Weir where large-scale dredging ceased in 1980. During the dry season, the weirs are effectively isolated water bodies when the river ceases to flow and any downstream impacts from dredging are contained by the barrages. This isolation was also extended between 1992 and 1997 as no water flowed over the barrages due to the El Nino reduction in summer rains. All sites, however, showed intra- and inter-seasonal changes in turbidity associated with changes in rainfall patterns, which are typical of tropical coastal streams. Significantly elevated aseasonal peaks in Black Weir, compared with Aplin Weir and Ross Dam (in 1998 and 1999 in Black Weir after this study), were most probably associated with localised gravel extraction (Appendix A, Figure A8 to A10).

2.3.6 Trophic status

Nutrient levels, primarily of nitrogen and phosphorus, may change seasonally with marked increases associated with rainfall events especially in summer. Inter-seasonal changes also occur with lower, but increasing levels of nutrients available in solution, particularly nitrogen, due to decreased rainfall and slower river flow rates (see Appendix A, Figures A14 to A23). The total nitrogen concentration in the Ross Dam, for example, increased after the 1990-92 El Nino event to a peak of 195.42 μg/l in 1996. It then decreased to 172.17 μg/l in 1997 and subsequently to an average of 77.86 μg/l in the following three years with the onset of another El Nino event (Appendix A, see Figure A22). Concentrations of phosphorus bound to sediment particles are enhanced by the dam and barrages that retain sediments within the river rather than being carried downstream and into Cleveland Bay. Heavy seasonal rainfall, land runoff and dredging can increase levels in the water column also by resuspension of sediments. Algal cells directly use the phosphorus bound to sediment particles resulting in algal blooms and the nutrient can then persist in suspension as it is rapidly recycled through a series of algal species.

Between 1994 and 2000, phosphorus concentrations remained consistently high with annual averages ranging between 10.04 and 20.33 μg/l in both weirs prior to and during the El Nino event (Appendix A, Figure A23). In contrast, concentrations in the dam increased to a similar level after 1997 (mean annual range: 10.83 to 15.16 μg/l), from significantly lower mean annual concentrations between 1994 to 1996 (range: 1.92 to 8.33 μg/l) with the onset of the heavy summer rains and increased water turbidity (Appendix A, Figure A24). For the Ross Dam, based on the above water quality data (1994-2000), monthly total nitrogen values ($N0_2 + NO_3 + NH_3$) exceeded 250 μg/l on several occasions in five out of seven years, although the mean annual values were less than 250 μg/l for all years (range: 63.92 to 195.42 μg/l). For the Aplin and Black weirs, the total N value exceeded 250 μg/l on only two sampling occasions (Black weir 94 (364 µg/l) and Aplin weir Aug 98 (366 µg/l)). For 1994-2000, the mean annual monthly total N for Black weir was 50.20 µg/l. The mean annual monthly total N for the Aplin weir was 57.49 µg/l respectively. Monthly phosphate values during this period were frequently above 5 µg/l and values ranged between 0 and 65 µg/l with a monthly average of 9.61 µg/l (sd. 9.56). For the Black and Aplin weirs, phosphorus levels were also frequently above 5 µg/l, with a range of 0 to 85 µg/l for Black weir (mean: 13.82 µg/l; sd. 10.69) and 0 to 45 µg/l for Aplin weir (mean: 12.51 µg/l; sd. 6.95).

These data concur with Finlayson and Gillies (1982) who described the Ross Dam as a mesotrophic water body. This classification was based on total nitrogen and phosphorus using the criteria according to King (1979) (Appendix A, Table A8). Accordingly, the Aplin and Black weirs may be described as between oligotrophic and mesotrophic status as nitrogen levels are predominantly below the 250 µg/l level while the phosphorus levels are predominantly above the 5 µg/l level. Between 1994 and 2000, nutrient concentrations of total nitrogen were consistently higher in the dam than the weirs and phosphorus for most years was higher in the weirs but not significantly different between locations. Based on Water Quality Guidelines for Queensland Waters (see Appendix A, Tables A6 and A7), the water quality for the Ross Dam and weirs was Good to Fair, with only total nitrogen levels for the dam rated as Poor.

It should be noted, however, that nutrient concentrations in the water might not reflect the true nutrient status of the water body. Nutrient levels may be influenced by plant metabolism and availability of nutrients in the sediments, which can act as a nutrient sink, especially for phosphorus. Nutrient input is variable and relatively low reflected in low algal and bacterial growth (particularly cyanophytes). However, the shallow nature of the weirs and dam margins and high levels of nutrients in the sediments has allowed prolific growth of macrophytes with occasional massive overgrowth that lower nutrient levels in the water column.

CHAPTER THREE

ESTABLISHMENT OF NON-INDIGENOUS FISHES IN NORTHERN QUEENSLAND

"*I must not, however, omit to mention one exotic fish, the acclimatisation of which in Queensland should receive our immediate and earnest attention the (Giant) Gourami…Why it has not long since been acclimatised is a puzzle that I have failed to solve*."

Mr D. O'Connor, Esq.,
Address to Royal Society of Queensland,
Nov 7 1896.

3.1 Introduction

3.1.1 Historical background: introductions of fishes into Australia and Queensland

An important objective of Naturalist and Acclimatisation Societies of the early colonial settlers was to encourage the introduction and establishment of exotic fauna and flora. Between 1850 and 1900, most of the eleven fish species introduced into Australia were into the southern colonies of New South Wales, South Australia, Victoria and Van Diemens Land (Tasmania). Fishes were selected often with the nostalgic belief that European and North American fishes possessed superior sporting qualities to the "depauperate" local fish fauna. In 1865, in a letter to the Fisherman's Magazine, London, Henry Francis Esq., writing under the pseudonym of "Anglicus Antipodeus", an "English gentleman not long in the Antipodes" declared: "There is no fly-fishing worthy of the name in New South Wales…the rivers afford one fish of great size and sterling excellence, the Murray Cod" (see Dunn 1991). Social class differences in the colonies were also reflected in the fish introduced: five salmonids (two salmon and three trout species) were considered as "game" fish suitable for the gentleman angler while Redfin perch, two Carp species, Tench and Roach were considered as "coarse" fish for the pursuit of the common working man. According to Dunn (1991), some colonists even viewed acclimatisation of fish, such as salmon and trout, as a conservation measure: to provide a safe haven for these species from the "increasing inroads of European civilisation." Goldfish and European carp were originally introduced for outdoor ornamental ponds (Shearer and Mulley 1978; Cadwallader and Backhouse 1983; McKay 1984), and reflected the increasing affluence and opportunities for leisure activity, such as landscaped gardening, of at least some members of colonial society.

In an address in 1896 to the Royal Society of Queensland, D. O'Connor, Esq., echoed the prevailing view that Queensland, like other colonies, possessed "very fine rivers, many of which are, however, remarkably poor, both in variety, in quantity, and in quality of the finny tribe." Earlier in July 1886, Mr O'Connor was responsible for the first non-indigenous fish introductions in Queensland with the release of 36 Redfin perch (*Perca fluviatilis*), 15 Trout (*Salmo* sp.), 3 European carp (*Cyprinus carpio*) and 3 Tench (*Tinca tinca*) into Gold Creek reservoir, near Brisbane, transported from Lake Wendouree, Victoria. While half the fish survived the epic journey, there were no subsequent reports of the establishment and fate of the Gold Creek reservoir fishes. According to Weatherley (1963), another unsuccessful attempt was made to introduce Redfin perch into the same reservoir. McCulloch (1929) also recorded Redfin perch in Queensland but does not provide any details on timing or location of the introduction. Of these initial species only Carp is known to have established elsewhere in Queensland. McKay (1984) noted an introduction of Carp in 1888 although no location or status of this population were given. Currently there are two genetically distinct Carp populations in southern Queensland: in the Murray-Darling system and in the Logan and Albert Rivers. The Murray-Darling population spread naturally and by human transport from NSW and Victorian populations in the 1960s (Broster 1995; Mallen-Cooper 1988; Mallen-Cooper *et al.* 1995), although the origins and timing of the introductions of the Logan-Albert populations are not known (Mackenzie *et al.* 2000). Bleakley and Grant (1954) listed the Rainbow trout (*Salmo gairdneri*) as a common freshwater fish in southern Queensland but did not give details on the location, timing of introduction or status of extant populations.

While game and coarse fish represented the first phase of introductions into Australia, a second phase, dominated by the introduction of aquarium fishes, began in the 1920s and has continued largely unabated. In the post World War 1 period, keeping exotic fish in aquaria became an increasingly popular past-time, and in 1924 became an organised hobby with the formation of the Aquarium and Terrarium Society of Queensland (P. Casey, ANGFA, Qld, *pers. comm.*). Fish were initially brought in on ships, often by sailors to supplement their income, from various ports of call around the globe. Whitley (1951) listed at least 54 genera of tropical fishes kept by enthusiasts in Australia during the early years of the hobby including four cichlid and nine cyprinodontid genera. The cichlids included members of the genus *Tilapia* and the live-bearing cyprinodontids included the Mosquitofish or Top Minnow (*Gambusia (affinis) holbrooki*), Guppy (*Poecilia (Lebistes) reticulatus*), Platy (*Xiphophorus maculatus*) and Swordtail (*Xiphophorus helleri*). According to Hamlyn-Harris (1929), the Mosquitofish, and the Guppy were, before 1928, already well known to "aquarium lovers" in Brisbane. The Mosquitofish was imported into New South Wales in 1921 due to its reputation as a predator of mosquito larvae in its place of origin, the south eastern States of the USA (Hildebrand 1919;

Barney and Anson 1921), and was released into creeks in the Sydney area in 1925 as a bio-control agent (Hamlyn-Harris 1929; McKay 1984).

Hamlyn-Harris (1929) reported that, in the Brisbane area, the Guppy, in spite of its lower tolerance to winter temperatures than Mosquitofish, was already being used for mosquito control in outdoor ponds and other water-holding structures. Successful trials of Mosquitofish from translocated NSW stock were conducted in evaporation tanks although there were no subsequent reports of releases into open waters in Queensland until 1941 when a widespread mosquito control program to combat malaria was begun by military personnel. The Mosquitofish was the first recorded non-indigenous fish species to be introduced into open waters in northern Queensland. These fish, from Brisbane stock, were translocated and then released in the Cairns area in 1943 and subsequently into other sites in northern Queensland with the added assistance of local shire councils and the State Health Department (McKay 1984).

In the post-war period the new waves of migration and economic growth in Australia were paralleled by a major growth in popularity of the aquarium hobby that gradually expanded into a multi-million dollar industry. There were also growing concerns that an unfortunate outcome of this popularity was the release of non-indigenous aquarium fishes into Australian waters and potentially negative impacts on indigenous aquatic communities (McKay 1977, 1986a,b,c; Arthington and Mitchell 1986; Arthington 1989). During the period from 1945 to 1994, at least 19 non-indigenous species (see Appendix 3, Table 3A) were introduced, most of which established breeding populations. A majority (79%) of these introductions were into Queensland waters, and at least 10 species (53%) were recorded in northern Queensland waters (see McKay 1978, 1984, 1987, 1989; Webb 1994). These fishes comprised four poeciliids (Guppy, Swordtail, Platy, and Sailfin Molly (*Poecilia latipinna*)) and six cichlids (Mozambique mouthbrooder, Spotted tilapia (*Tilapia mariae*), Jewel cichlid (*Hemichromis guttatus*), Oscar (*Astronotus ocellatus*), Green severum (*Heros severus*) and Midas cichlid (*Amphilophus citrinellum*)).

The poeciliids (Guppy, Swordtail, Molly and Platy) were probably the first to be introduced, but the timings of the introductions are uncertain. McKay (1978) reported that these poeciliids, along with Mosquitofish, were widespread in southeastern Queensland creeks, and, in the first surveys of the regions, in several creeks in central and northern Queensland. The Platy was found at Babinda and guppies were abundant in Cairns, Innisfail and Ingham. The second group of fishes, the cichlids, began to appear in Australian waters in the late 1970s with the first (Convict cichlid (*Archocentrus nigrofasciatum*), Spotted Tilapia and the Jack Dempsey

(*Archocentrus octofasciatum*)) reported in 1978 from a thermally polluted creek downstream from a power station in Victoria (Cadwallader *et al.* 1980). McKay (1977) reported that tilapiine species, declared noxious (i.e., prohibited) in Queensland, were available from some aquarium retailers in Brisbane, and were also maintained by aquarists in outdoor ponds. The subsequent release and dispersal in Queensland during the 1980s and early 1990s of the Mozambique mouthbrooder and other cichlids has been well documented (e.g., McKay 1984; Arthington *et al.* 1984; Webb 1994). Since 1994, several fish surveys have been conducted in northern Queensland, including Cape York Peninsula, that have provided distribution data on non-indigenous fishes (e.g., Webb *et al.* 1997; Herbert and Peeters 1995; Pusey and Kennard 1996, Russell *et al.* 2000).

3.1.2 Success of non-indigenous fishes: risk assessment

Mack *et al.* (2000) noted the increasing recognition by some governments that the traditional approach has been inadequate to stem the tide of such introduced organisms and that there was a need for policy changes. However, risk assessment of potentially invasive species is still a developing field and protocols have been implemented in relatively few countries, including Australia. These protocols, used by the Australian Quarantine Service (AQIS), are mainly for screening imported plant species and products, but also bio-control agents such as invertebrates, fungi and microorganisms (Pheloung *2001*). These protocols are now gradually being extended to vertebrates including birds (Duncan *et al.* 2001; Bomford and Sinclair 2002), mammals (M. Bomford *pers. comm*), and to fish. Arthington *et al.* (1999) provided a report towards the development of an Ornamental Finfish Import Risk Assessment scheme by AQIS (Kahn *et al.* 1999). While the focus of the AQIS report was on the ecological and economic risk of pathogens entering Australia with host fish, the contributory report by Arthington and co-workers assessed the probability of non-indigenous fish surviving and establishing self-maintaining populations in Australian waters. They categorised non-indigenous fish species according to records of establishment of populations outside their natural range in Australia and abroad, as well as their taxonomic affinities. Non-indigenous fish that were already established, present and not established, or present but of unknown or uncertain status were regarded as species with a very high probability of establishment in Australia. All other species in the same genus with a history of successful establishment in Australia and elsewhere were regarded as species with a high probability of establishment. All non-indigenous fish in families (other than those with representatives established in Australia) that contain species which have established successfully elsewhere were regarded as species with a moderate probability of establishment. All other species that did not fit into these categories were regarded as species with a residual probability of establishment.

Cassey and Arthington (1999) provided an empirical model of introduction success at the individual stages of invasion, arrival, establishment and persistence. Based on this model and a review of relevant literature, they proposed that previous success at invasion, propagule pressure and habitat matching were reasonably robust predictors of invasion success. These researchers and others (e.g., Williamson 1996; Moyle and Light 1996; Brown *et al.* 2000) also noted that key attributes of successful invaders were wide physiological tolerance limits, broad habitat requirements and behavioural flexibility, especially in relation to feeding strategies including omnivory and opportunistic predation. Such hardy generalists, therefore, given sufficient founder numbers (that may theoretically even be a single female in the case of livebearers or incubating mouthbrooders) are likely to establish and persist due to a high probability of encounter with a suitable environment, particularly if it is similar to the one in which they evolved. There are many examples, however, where non-indigenous fishes have failed to establish in some or all introduction sites (see Welcomme 1992; Baltz and Moyle 1993; Brown *et al.* 2000). Based on the concept of ecological "resistance” to invasion proposed by Elton (1958), Baltz and Moyle (1993) provided two general hypotheses: the environmental and biotic resistance hypotheses. For the former, one or several abiotic factors may be limiting, such as the absence of suitable spawning substrate or unfavourable flow regimes. For the latter, the introduced fishes cannot break into an established community structured by strong biotic interactions. Separating the limiting effects of each may be difficult; while environmental resistance may be easier to observe directly, the importance of biotic resistance may be indirectly observed in highly disturbed habitats where indigenous communities have been compromised by anthropogenic activity (see Chapter 4).

In recognition of the difficulties of making accurate predictions, Cassey and Arthington (1999) suggested that three factors – previous success at invasion, propagule pressure and habitat matching – could be used as qualitative guidelines in the initial phase of risk assessment. They also noted that these guidelines were not directed at predicting ecological impacts but were concerned only with predicting the fate of introduced species.

The main focus of this study was to provide a comprehensive assessment of the current number, distribution, and the rate and mode of dispersal of non-indigenous fishes in northern Queensland waters, and to assess the success of these non-indigenous fishes in establishing breeding populations. This was done by use of absolute latitudinal range overlap as a measure of habitat matching, the number of known introductions as a measure of propagule pressure, in combination with ecological tolerance data (from the literature) and residency time in northern Queensland waters. These data were considered in the national context in relation to the numbers of non-indigenous fishes estimated in captivity or imported into Australia and the species richness of the indigenous freshwater fish fauna. These introductions were considered in a global context: is the establishment of exotic fishes in northern Queensland following global trends?

3.2 Methods

Surveys were conducted at freshwater sites in northern Queensland on the coastal plain between the Burdekin and Daintree river catchments and on the Atherton Tablelands (Figure 2.2). Sampling methods included fish traps, seine and gill nets, portable and boat-mounted electro-shockers and direct observation with binoculars from the water's edge or underwater with face mask and snorkel. The position coordinates (latitude and longitude) for each site were obtained with a GPS unit. Data on non-indigenous fish provided by other researchers within the region during the study period were included in distribution maps. Summary details of these sites are included in Appendix B, Table B1.

Data on the range of non-indigenous fishes recorded in northern Queensland waters and elsewhere overseas were obtained from databases (FishBase; United States Geological Survey: Non-indigenous Aquatic Species) and various literature sources (e.g., Staeke and Linke 1985; Wooten *et al*. 1988; Axelrod 1993; Conkel 1993). Available data on physiological tolerances (temperature, dissolved oxygen and salinity) of non-indigenous species in northern Queensland were also obtained from the literature. Historical records of introductions of these species in northern Queensland were obtained from surveys previously mentioned (section 3.1.1) and records from J. Johnson, Ichthyology Section, Queensland Museum, Brisbane (*pers. comm.*).

As a measure of habitat matching, indigenous and non-indigenous range comparisons were expressed as a percentage of absolute overlap between the two ranges, using latitudinal range irrespective of hemisphere (N or S). Tropical northern Queensland was defined as between 10.4º S and 23.2º S, from the tip of Cape York to Rockhampton (Tropic of Capricorn). In virtually all cases, there is no information available on the number of individuals of each species introduced at each site. Although also imprecise, the number of known introductions of non-indigenous fishes in northern Queensland was used as a measure of propagule pressure. Residency times were based on the first reports available of each non-indigenous species in northern Queensland waters since, for the majority of species, data on actual times of establishment of breeding populations are not available.

3.3 Results

3.3.1 Distribution of non-indigenous fishes in northern Queensland

Non-indigenous fish were recorded at 158 freshwater sites between the Burdekin and Daintree River catchments (Figures 3.1 to 3.2) and at a further 44 sites from other recent surveys (see Appendix B, Table B1 for references and site details). Seventeen non-indigenous fish species (10 cichlids, 5 poeciliids, 1 cyprinid and 1 belontiid) were recorded during the study from open waters in northern Queensland (Table 3.1 and Appendix B, Figure B1 to B18). These include four species (Three-spot gourami (*Trichogaster trichopterus*), Burton's haplochromis (*Haplochromis burtoni*), Green terror (*Aequidens rivulatus*) and Firemouth cichlid *(Thorichthys meeki*)) not previously recorded in open waters in Australia. The Three-spot gourami was first reported in 1998 from a sugar-cane irrigation channel and subsequently from freshwater lagoons associated with Sheep Station Creek in the lower Burdekin region (C. Perna, *pers. comm*.). These sites were characterised by low fish species diversity and dense mats of introduced floating vegetation, especially the Water hyacinth. The Gourami had established breeding populations and was common in the sites surveyed. A second non-indigenous species, the Mosquitofish, was abundant at these sites. Reports in 2002 of the Three-spot gourami in the Ross River catchment were confirmed in February 2003 with adult fish observed in the lower reaches of Campus Creek above its confluence with the Aplin Weir section of the Ross River. In March 2003, an approximately 4 kg specimen of a cyprinid was found among a large number of dead fish in Gleeson's weir, Ross River, presumably killed by anoxic conditions in the weir created by bacterial decomposition of plant material and a lack of flushing of the weir. It was identified as a cyprinid, possibly a wild variant of the European carp, or a member of the African-Asian carp genus, *Labeo*, although full identification was not possible. This is the first report of a cyprinid from open waters in northern Queensland. A subsequent survey in the weir by QFS Fisheries personnel using a boat-mounted electroshocker failed to detect any further specimens. While this particular fish may have been an isolated introduction in the Ross River, there is still a possibility that other specimens may be detected in one of the weirs.

There are also new site records for all of the twelve other non-indigenous species. Of these, two species (the Sailfin molly and Convict cichlid) have not previously been recorded in northern Queensland open waters. There are eight new species records for the Ross River catchment (Convict cichlid, Jewel cichlid, Green severum, Burton's haplochromis, Green terror, Firemouth cichlid, Three-spot gourami and the unidentified cyprinid sp. A). Juveniles and fry of two of these species (Jewel cichlid and Burton's haplochromis) were observed in small streams in the

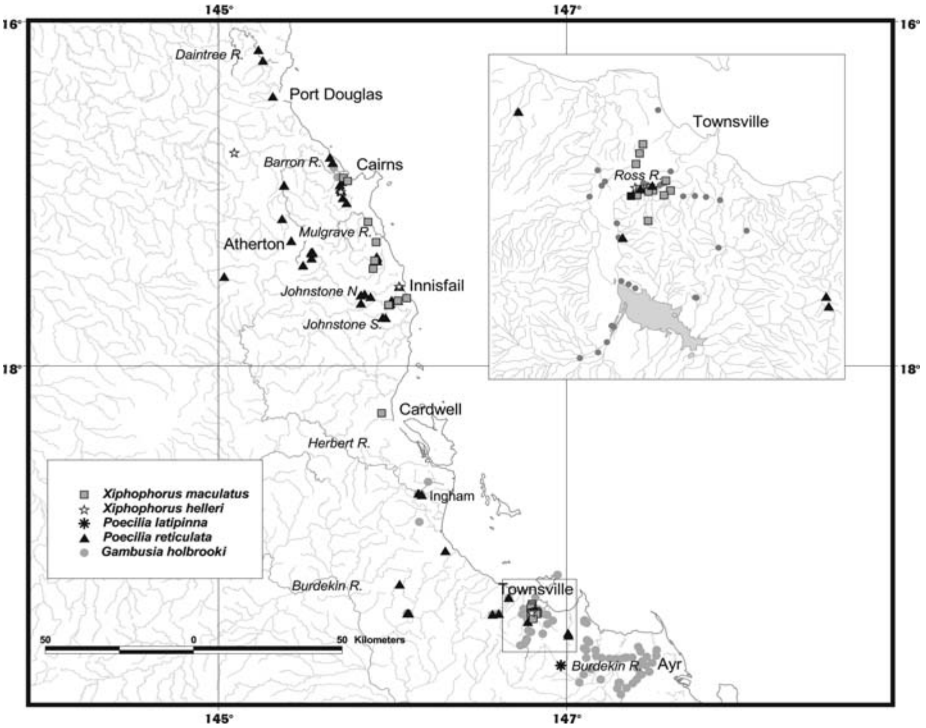


Figure 3.1 Distribution of non-indigenous poeciliid fish species in northern Queensland

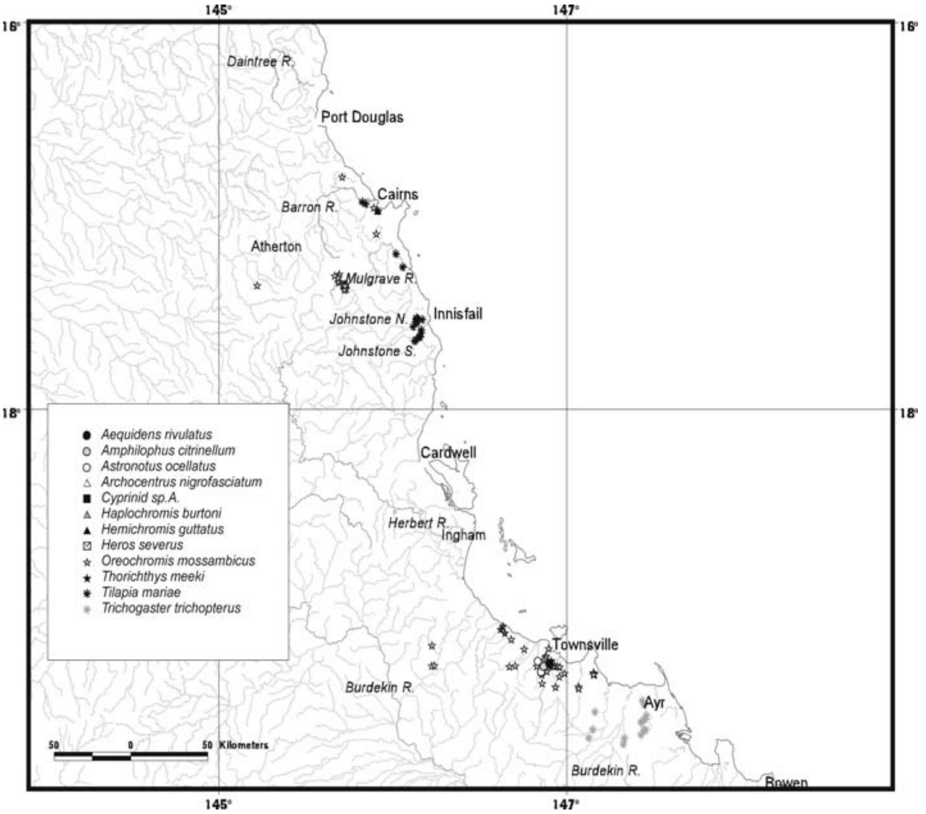


Figure 3.2 Distribution of non-indigenous cichlid and other fish species (excluding poeciliids) in northern Queensland

catchment and in small side channels formed by islands in the main river channel. These side channels were seasonally isolated from the main channel by dense patches of vegetation, including Paragrass and Water hyacinth (Chapter 4).

Of the fifteen species of non-indigenous fishes recorded from the Ross River catchment, nine species have now established breeding populations. Introduced species now represent about 40% of the total fish species (not including catadromous forms unable to migrate upstream due to the barrages). Furthermore, about 40% of all introduced freshwater fish in Australia have now been recorded from northern Queensland waters. Excluding the unidentified cyprinid, 11 of the 16 species have established breeding populations and seven species were recorded from three or more sites. There was a strong correlation ($r = 0.894$; $p = 0.041$) between the frequency of known introductions and successful establishment of a breeding population (Figure 3.3). All species with a frequency of known introductions equal to, or greater than 3.0 had established breeding populations. However, three species with lower known introduction frequencies (Three-spot gourami, Jewel cichlid and Burton's haplochromis) had also established breeding populations. The most frequently collected species, as a percentage of total sample sites were Mosquitofish (54%), Mozambique mouthbrooder (40%) Guppy (26%), Spotted Mangrove cichlid (10%) and Platy (11%) (Table 3.1). Excluding the Black spotted mangrove cichlid, these species had the widest introduced range between 2.4 and 2.9° latitude (Table 3.1) and all have residency times greater than 25 years. There were marked differences in the regional distribution of the dominant poeciliids (Guppy and Gambusia) and cichlids (Mozambique mouthbrooder and Black-spotted mangrove cichlid). The Guppy was observed at 45% of sites north of Ingham, but only 12% of sites to the south. In contrast, the Mosquitofish was observed at only 3% of sites to the north of Ingham, but was collected from 90.5% of sites to the south. The Mozambique mouthbrooder and Black-spotted mangrove cichlid were collected from a similar percentage of total sites north of Ingham (28% and 26% respectively), but only Mozambique mouthbrooder was observed south of Ingham (48% of all sites surveyed). North of Ingham, the Mozambique mouthbrooder was the most frequently observed cichlid on the Atherton Tablelands, while the Black-spotted mangrove cichlid was predominant in coastal sites.

Excluding the Platy, Swordtail and Three-spot gourami, the majority of species had narrow ranges in open waters and were observed in less than 3% of all sites surveyed. Eight of these species were observed only in the Ross River catchment (Green sevurum, Firemouth cichlid, Green terror, Burton's haplochromis, Convict cichlid, Midas cichlid, Oscar and Jewel cichlid) The Platy was found at several sites between Cairns and Townsville where populations varied from low to high abundance. Small populations of the Swordtail were found at one site in the

Ross River catchment and at two sites in the Cairns region. The species was also found along with Guppies in larger numbers at a site near Innisfail in the ponds of a crocodile farm and in adjacent stormwater drains. According to staff, these fish had been stocked to control mosquito larvae. In the Cairns region, Black-Spotted mangrove cichlid, Platy and Jewel cichlid were found in ornamental ponds in the Central Business District, while the Mozambique mouthbrooder, Mosquitofish and Platy were found in Centenary Lake, a large freshwater impoundment in public gardens. Swordtails were observed immediately downstream in a creek which flowed out of a wildlife park in a northern Cairns suburb which was also reported by Lear (1987) to be the original site of introduction of tilapia in the Cairns region. Recently, fisheries officers confirmed the presence of the Midas cichlid in resort ponds at Port Douglas, north of Cairns, and Koi carp and feral Goldfish (*Carassius auratus*) in a large private impoundment in the Mulgrave River catchment to the south of Cairns (J. Russell, Northern Fisheries Centre, Cairns, *pers. comm*.). For the Ross catchment, three cichlid species, the Convict cichlid, Firemouth cichlid and Burton’s haplochromis, were also first found in ornamental ponds and then in a nearby creek. The Mozambique mouthbrooder was found in farm dams at Oak Valley to the west of Townsville, at Cungulla to the south of Townsville and, at the latter location, in stormwater drainage channels and a creek close to the dams. While Oscars have previously been reported from Freshwater Creek in Cairns, none were found in the Cairns region during this survey. However, the species was reported from the Bohle River and was collected in small numbers from the Aplin and Black Weirs of the Ross River.

The five species for which no evidence of established breeding populations was observed (Green Terror, Firemouth, Green severum, Convict cichlid and Sailfin molly) all had a frequency of known introductions less than 3.0 and residency times of less than about five years. These species were found at no more than two sites and five or fewer individuals were collected at these sites.

3.3.2 Establishment success of non-indigenous fishes in northern Queensland

There was a positive correlation between the previous history of introductions, the frequency of introductions (index of propagule pressure) and successful establishment of breeding populations in northern Queensland waters (Table 3.1 and Figure 3.3). Fourteen out of 16 species (87.5%) had previously been introduced into other countries. Only 50% of the species that had been previously introduced into four or fewer countries have currently established breeding populations while 87.5% of those species previously introduced into five or more countries have established breeding populations. Of the two species not previously introduced elsewhere, Burton’s haplochromis and the Green terror, only the former has established a breeding population, while the latter was reported only as a single specimen. From Table 3.1, excluding the unidentified cyprinid, 12 species (75%) had effective range overlaps with the latitudinal range for tropical northern Queensland. For several species, notably the Jewel cichlid, Burton's haplochromis, Green terror and Green severum, there is little or no ecological tolerance data available in the literature. However, at least 11 species (69%) had wide tolerances for salinity, high temperature or low oxygen concentration, or for a combination of these factors. Of nine non-indigenous species with an effective range overlap of greater than 40 percent, seven species had established breeding populations. The other two species, the Convict cichlid and Firemouth cichlid, most probably very recent introductions, were found at two sites and one site respectively. Only one individual Convict cichlid was collected at each site and five Firemouth cichlids at the single site.

Of seven species with a range overlap between 0 and 40%, four had established breeding populations (Burton's haplochromis, Jewel cichlid, Oscar and Mosquitofish). The first three species had no range overlap and had an equatorial native range, while the Mosquitofish, also with no range overlap, is a eurythermal species with a temperate to subtropical indigenous range (eastern USA from New Jersey south to Florida). Large numbers of each of these species were recorded at sampling sites, with all species, except Burton's haplochromis (one location), recorded from three or more sites. The other three species in this category for which no breeding populations were observed were recorded from two or fewer sites and four or less individuals collected (Sailfin molly: one site – four individuals; Green terror: one site – one individual; Green severum: two sites – one individual per site).

Five of the six known euryhaline species recorded in open fresh waters (Mozambique mouthbrooder, Black-spotted mangrove cichlid, Guppy, Mosquitofish and Midas cichlid) had established breeding populations. The Sailfin molly, also euryhaline, is a mainly sub-tropical species with a relatively low range overlap (20%) with tropical northern Queensland, but with a low known frequency of introductions (1) in the region. All seven species with hypoxia tolerance (Mosquitofish, Guppy, Platy, Swordtail, Mozambique mouthbrooder, Oscar, Three-spot gourami) had established breeding populations. Seven of the nine species with high temperature tolerance (Mosquitofish, Guppy, Platy, Swordtail, Mozambique mouthbrooder, Black-spotted mangrove cichlid and Three-spot gourami) had established breeding populations. Two species with high temperature tolerance, the Sailfin molly and Convict cichlid were found at only one and two sites respectively and only in very small numbers. Of the four species for which no ecological tolerance data are available (all cichlids: Jewel cichlid, Burton's haplochromis, Green terror and Green severum) the first two have established breeding populations.

Table 3.1 Non-indigenous freshwater fishes recorded in tropical northern Queensland, their indigenous and introduced range and absolute percentage range overlap, number of known introductions (*f*), breeding status (🐟 = established breeding populations) and their general ecological tolerances (Salinity, Temperature [High] and Oxygen concentration [Low]), and % oc. = percentage occurrence in all sites surveyed

Family	Species	Indigenous Range (I.R.)	I.R. ºLat.	A.L.R Overlap (%)	Tolerant sp. S, T, O	🐟	Prev. Number Intros	Intro *f*	% oc	Introduced range (ºLat in NQld)
Poeciliidae	***Gambusia holbrooki***	40ºN-25ºN	15	0	S, T, O	🐟	19	>5	54.0	2.93
	Poecilia reticulata	18ºN-5ºN	13	58.5	T, *O*	🐟	46	>5	26.2	2.91
	P. latipinna	31ºN-21ºN	10	22.0	S, T		12	1	0.005	2.55
	Xiphophorus maculatus	19ºN-15ºN	4	100	T, *O*	🐟	15	>5	10.9	2.44
	X. helleri	20ºN-15ºN	5	100	T, *O*	🐟	24	>5	1.5	-
Cichlidae	*Oreochromis mossambicus*	35ºS-13ºS	22	46.4	S, T, O	🐟	84	>5	39.6	2.69
	Tilapia mariae	15ºS-5ºS	10	46	S, T	🐟	3	3	9.9	0.73
	Thorichthys meeki	20ºN-15ºN	5	100	S		4	1	0.005	-
	Hemichromis guttatus	7ºN-2ºN	5	0	?	🐟	4	3	0.01	-
	Haplochromis burtoni	9ºS-3ºS	6	0	?	🐟	0	1	0.01	-
	Astronotus ocellatus	14ºS-8ºN	22	16.4	O	🐟	6	3	2.0	-
	Aequidens rivulatus	4ºS-0.5ºS	3.5	0	?		0	1	0.005	-
	Heros severus	14ºN-1ºN	13	27.7	?		1	2	0.005	-
	Archocentrus nigrofasciatum	15.7ºN-9ºN	6.7	79.3	T		4	2	1.5	-
	Amphilophus citrinellum	12.5ºN-10ºN	2.5	72.0	S	🐟	2	1	3.0	-
Belontiidae	*Trichogaster trichopterus*	16.5ºS-8.2ºS	8.3	73.5	T, O	🐟	7	2	7.4	0.42
Cyprinidae	Cyprinid sp. A	-		-	**-**	?	?	1?	0.005	-

Environmental tolerances of non-indigenous fishes: Maruyama 1958; Allanson and Noble 1964; Potts *et al.* 1967; Burggren 1979; Whitfield and Blaber 1979; Peer 1981; Kramer and Mehegan 1981; Chervinski 1982; Peer and Kutty 1982; Weber and Kramer 1982; Kasim 1983; Kyle 1984; Behrends and Smitherman 1984; Chung 1985; James 1985; Stauffer 1986; Siemien and Giusto *et al*. 1998; McKinsey and Chapman 1998; McManus and Travis 1998; Muusze *et al*. 1998; Williams *et al*. 1998; Bailey *et al.* 1999; Winckler and Fidhiany (1999); Almeida-Val *et al.*(2000); Cnaani *et al.* 2000; Prodocimo and Freire 2001; Chung 2001; Ikenga *et al*. 2001; Morgan and Gill 2001; Poulakis *et al*. 2002

Home Distribution of non-indigenous fishes: Trewavas 1983; Staeck and Linke 1985; Wooten *et al*. 1988; Lydeard *et al*. 1995; Wischnath 1993; Conkel 1993; Linke and Staeck 1994; www.fishbase.org.2003

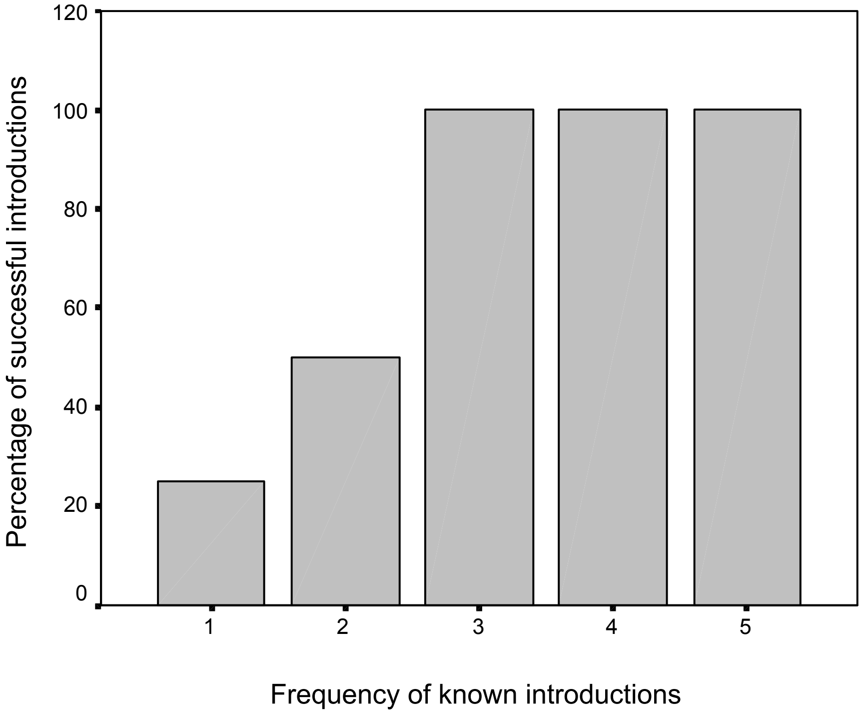


Figure 3.3 Percentage of non-indigenous fish species with established breeding populations in relation to known frequency of introductions in northern

The last two species were only reported from one and two sites respectively and only as single individuals.

3.4 Discussion

3.4.1 Distribution of non-indigenous fishes in northern Queensland

Introduction and spread of non-indigenous fishes in northern Queensland is continuing with at least four species (excluding the unidentified cyprinid) new to Australian open waters and new site records and range extensions for a number of already established species. The total number of species now reported from northern Queensland waters is 17, which represents a 70 percent increase since 1994 when 10 species were recorded (Webb 1994). Non-indigenous fishes were present in all sites visited during the study and recent surveys conducted by other researchers (e.g., Russell and Hales 1997; J. Russell, Northern Fisheries Centre, Cairns, *pers. comm.*) found that there are few major river catchments where exotic fish species have not established. In contrast, for Cape York Peninsula north of the Daintree River, where human population is sparse and urban and rural development is limited, no exotic species were reported during recent surveys of the major river systems (Herbert and Peeters 1995), although there have been a small number of reports of introductions in the region. In 1989, Tilapia (probably the Black-spotted mangrove cichlid) was reported from Cooper Creek and Hutchinson Creek, which are part of the northern Daintree catchment, although subsequent surveys failed to detect them (A. Hogan, DNR, Atherton, *pers. comm.*). In August 2003, a specimen of the Mozambique mouthbrooder was collected from the Endeavour River at Jensen’s Crossing, although it is not known if this specimen was an isolated introduction (J. Russell, Northern Fisheries Centre, Cairns, *pers. comm.*). There have also been reports of Guppies in small streams in the Weipa area although these have not been confirmed (B. Pusey, *pers. comm.*).

The Ross River catchment now contains 15 of the 17 species recorded for northern Queensland. Compared with the six species recorded for the catchment in 1994 (Webb 1994). In comparison, all other river systems in Queensland that have been surveyed have fewer introduced, non-indigenous species. In northern Queensland, the Burdekin, Daintree and Mossman Rivers each have one introduced species, which is about 2% of their freshwater fish faunas (Russell *et al.* 1998; B. Pusey, *pers. comm.*), the Barron River has six introduced species, about 12% of its freshwater fish fauna (Russell *et al.* 2000), and the Johnstone River catchment has three species, about 8% of its freshwater fish fauna (Russell and Hales 1993). In southeastern Queensland, the Brisbane River has six species, which is about 14% of its freshwater fish fauna (McKay and Johnson 1990; J. Johnson, Ichthyology Section, Qld Museum, *pers. comm.*). Australia’s largest river system, the Murray-Darling, which drains an area of 265,000 km^2 through southern Queensland, New South Wales, Victoria and South Australia, is many times larger than the Ross River, has more indigenous fish species but has fewer (10) non–indigenous species, representing about 26% of its freshwater fish fauna (MDBC 1988).

Introductions of new species are also continuing to occur elsewhere in Australia. The most recent confirmed introduction into Australian waters was the White Cloud Mountain minnow, *Tanichthys albonubes,* reported in November 2002. This species is indigenous to mountain streams in the Guandong Province, China, and has established a breeding population, near Gosford, in New South Wales (T. Rayner, *pers. comm*.). The total number of introductions into Australian waters, therefore, is now at least 36 species (see Appendix B, Table B2,). Forty-three per cent of all non-indigenous fishes reported in Australia and 48% of all non-indigenous species with established breeding populations occur in northern Queensland waters. From the historical record (Appendix B, Figure B19) the rate of introductions in Australia has been accelerating since the 1970s and especially in the last ten years. A similar, virtually exponential growth has occurred during the mid to latter part of the 20th century in industrialised countries in lower latitudes where the aquarium hobby became an economically important industry during this period. In these countries, tropical or sub-tropical ornamental species now represent the largest proportion of introductions into open waters (Appendix B, Figure B19 to B22). In other countries where introductions have occurred primarily to establish fisheries (Appendix B, Figures B23 to B26), the rate of introductions has been much lower, although in some countries, such as Spain, the growth during the latter half of the 20th century in the number of fish species introduced for recreational fishing has also been exponential (Elvira and Almodovar 2001).

Australia currently has a much smaller proportion of non-indigenous fishes in comparison with some locations where non-indigenous fishes now dominate the aquatic fish fauna, e.g., Florida and Hawaii (see, Appendix B, Table B3). The number of species with established populations in Australia (23) represents about 10% of the total Australian freshwater fish fauna. As the ornamental fish trade is based largely on tropical species, there is a high probability that Queensland waters, particularly in northern tropical Queensland, will be at significant risk of further introductions and catchments being dominated by non-indigenous fishes, as the current situation in Ross River suggests.

The current distribution of previously established non-indigenous fishes in comparison with earlier surveys (see Webb 1994) indicates that localised natural dispersal is continuing, probably associated with wet season flooding. The Mozambique mouthbrooder, for example, having dispersed downstream from its original point of introduction in the Upper Barron catchment into

Tinaroo Dam, has recently been reported further downstream below the dam at Mantaka, presumably having survived the spillway descent (H. Kuhn, Mitchell River Catchment Management Group, Mareeba, *pers. comm.*). Also, this species was found in Leichardt Creek and the Christmas/Log Creek system, 11 km and 8 km, respectively, north of Bluewater Creek in Thuringowa that was previously the northern limit of the species in the Townsville-Thuringowa region (Webb 1994). While dispersal could have occurred over such a distance by interconnected stormwater ditches in the area, fish probably dispersed by coastal "creek-hopping" assisted by northerly longshore drift and possibly by lowered inshore salinity after heavy monsoonal rains, although the species is euryhaline and can tolerate gradual acclimation to normal seawater.

Significant changes in range are continuing beyond local dispersal. These extensions are primarily by human translocation (e.g., new site locations for the Jewel cichlid and Oscar in the Townsville-Thuringowa area, for both tilapiine species in the Upper Barron River catchment, Atherton Tablelands, and for Mozambique mouthbrooder in farm dams at Oak Valley and in farm dams and nearby Barrambush creek, Cungulla, to the south of Townsville) (Appendix B, Table B1). The adult Mozambique mouthbrooder, collected from Jensen's Crossing on the Eneavour River, Cooktown, represents a significant increase in the northern part of the species' range. This specimen may have been used for livebait fishing and brought to Cooktown, possibly from Cairns (J. Russell, QFS, Northern Fisheries Centre, Cairns, *pers. comm.*). Such translocations provide an opportunity for further localised dispersal following release. It is evident from the survey that impoundments such as public and private ornamental ponds and farm dams continue to be important introduction sites of these exotic species. Virtually all of these water bodies that contained non-indigenous species or were the probable source of introductions into open waters, were unsecured and below ground level. Such waterbodies are clearly introduction "hotspots", especially in tropical or flood-prone regions, and there is a need for urgent review of legislation at both local and State level regarding the construction and use of such impoundments on both public and private land.

The non-indigenous fish fauna in northern Queensland continues to be dominated by cichlids and poeciliids, although the first species from other families (Cyprinidae and Belontiidae) have now been reported. The Three-spot gourami (Suborder Anabantoidei, Family Belontiidae) is now well established in two creek systems in the lower Burdekin area and will most probably establish breeding populations in the Ross River. The introduction of Three-spot gouramis in the Ross River catchment probably occurred within the last two years. The size and reproductive status of this population are not known. This species is originally from South-East Asia, (Kottelat 1985, 1998; Allen 1991; Kottelat *et al.* 1993) where it often occurs in heavily vegetated, shallow, sluggish or standing water and in seasonally flooded habitats. This species and other anabantoids, such as the Climbing gourami, *Anabas testudines,* and Giant gourami, *Osphronemus goramy,* are of commercial importance as food fish in parts of South East Asia, which resulted in their introduction into other countries within the region, such as the Philippines (Juliano *et al*. 1989). Members of this group are also very popular aquarium species and have been exported and cultured worldwide. The circumstances of the original introductions in northern Queensland of the Three-spot gourami are unknown, although it is probable the fish were aquarium discards. This species has been released or escaped into open waters and established feral populations in several countries, including the USA (Florida) (Courtenay *et al*. 1984), Colombia (Welcomme 1988, Taiwan (Liao and Liu 1989), Sri Lanka (Welcomme 1988), Namibia (FAO 1997) and Papua New Guinea (West and Glucksman 1976). Very little information is available on how most of these fish entered open waters although, in the Dominican Republic, Lever (1996) reported that Three-spot gouramis escaped from a fish farm into the Rio Ozama during a hurricane and the species has now become a dominant component of the fish fauna.

Like other members of the Anabantoidei, the Three-spot gourami is remarkable in possessing an auxiliary respiratory structure, the labyrinth organ associated with the gills, that enables the species to switch rapidly between water breathing and air-breathing depending on available oxygen in the water (Burggren 1979; Heisler 1993; Berra 2001). This adaptation makes it ideally suited to the habitat conditions of many northern Queensland rivers, creeks and associated lagoons such as those in the lower Burdekin region which have been highly modified for crop irrigation and by the invasion of exotic vegetation including Paragrass and Water hyacinth. The dense vegetation combined with the high organic material and nutrient loads results in very high biological oxygen demand (BOD) with diurnal and seasonal hypoxic to anoxic conditions created within these systems. Lowered dissolved oxygen levels can result in reproductive failure especially of benthic spawners, due to adverse effects on eggs and larvae (Sieffert *et al.* 1973; Dombeck *et al.* 1984) and occasionally, especially in the tropics, the mass mortality of adults (Townsend *et al*. 1992; Sergeant and Galat 2002).

Such adverse conditions have probably favoured the establishment and proliferation of the Three-spot gourami and the Mosquitofish at the expense of indigenous species less able to tolerate hypoxic conditions. Recent surveys found that these non-indigenous species were common to abundant in lagoons associated with Sheep Station Creek. Only five indigenous species were recorded in these lagoons and all were very low in abundance (Perna 2003). These lagoons were overgrown with floating mats of Water hyacinth where oxygen concentrations frequently were below levels lethal to many native species. The indigenous Empire gudgeon

(*Hypseleotris compressa*) (*pers. obs.*) and the poeciliids, such as the Mosquitofish, are able to survive in stagnant water by utilising the oxygen in the thin surface water layer and by air gulping (Cech *et al.* 1985). Of the other indigenous species, the predatory Tarpon has a modified swim bladder that is highly vascularised and enables it to breathe in open patches of stagnant water also by gulping air at the surface (Geiger *et al.* 2000).

There is virtually no information on the potential ecological impacts of the Three-spot gourami in its introduced range. The species is carnivorous feeding opportunistically on zooplankton, crustaceans and terrestrial and aquatic insects (Rainboth 1996). It is territorial and aggressive and, according to Liao and Chiu (1989), was strongly suspected, as a resource competitor, to have caused damage to Taiwanese populations of the endangered Chinese barb, *Puntius semifasciolata.* The Three-spot gourami is likely to disperse locally, beyond its current distribution within the upper reaches of Sheep Station Creek and Barrattas Creek aided by the network of irrigation channels and seasonal flooding within the catchment. The recent occurrence of the species in the Ross River catchment provides further opportunity for the species to disperse and, given its tolerance to low oxygen and high temperatures, to establish self-maintaining populations in northern Queensland.

The discovery of a dead cyprinid, possible either a European or Asian carp in the Ross River is cause for concern in view of the history and subsequent spread of the former species in Australia (see Brumley 1996) and the latter species being well adapted to tropical conditions as a member of the fish fauna of the Indian subcontinent (Jayaram 1981). This is the first report of a cyprinid in open waters in northern Queensland, although there have been reports of ornamental carp in large lakes in the Cairns and Mulgrave regions (J. Russell, QFS Northern Fisheries Centre, Cairns, *pers. comm.*). The discovery of the Asian tapeworm, *Bothriocephalus acheilognathi*, introduced initially into Australia with European carp and found in the Guppy and Mozambique mouthbrooder in Wright Creek, may indicate that there are, or have been, feral cyprinid populations either in this creek, adjacent creeks or nearby waterbodies containing infected fish (see also Discussion section Chapter 6).

The two most widely distributed, non-indigenous species in northern Queensland are the two poeciliids, the Guppy and Mosquitofish, and these show interesting distribution patterns. The former is the predominant poeciliid species in coastal fresh waters north of Ingham and inland on the Atherton Tablelands, while the latter is the predominant species south of Ingham. The Mosquitofish was not recorded in surveys on the Atherton Tablelands. As previously noted, the Mosquitofish was introduced into northern Queensland during the 1940s for control of mosquito larvae. The absence of Mosquitofish and widespread occurrence of Guppies on the Atherton

Tablelands, where the largest concentration of Allied troops were stationed, suggests that Guppies may have been used in this wartime mosquito control program. Hamlyn-Harris (1929) considered the Guppy to be far more suitable for use for biological control in the tropics than the Mosquitofish because of the greater climate match for the former species. *A*t this time, the Guppy was introduced elsewhere in the tropics, e.g., Singapore, (Chou and Lam 1989) and later into several other countries for mosquito control (Welcomme 1988). In 1958, State Fisheries personnel introduced the Guppy into a large lagoon at Nhullumby, in the Northern Territory, Australia, to control mosquito larvae (P. Casey, ANGFA, Qld., *pers. comm.*). Farm employees introduced them more recently for the same purpose, along with Swordtails, in the ponds of a crocodile farm near Innisfail.

3.4.2 Establishment success of non-indigenous fishes in northern Queensland

Of the known introductions of non-indigenous fishes in northern Queensland, 11 species (71%) have successfully established breeding populations. Eight of these species have been present for more than 10 years. Of the recent introductions (less than 10 years) four out of nine species (44%) have established breeding populations. The breeding status of five species reported since 1997 (Green terror, Green severum, Firemouth cichlid, Convict cichlid and the unidentified cyprinid sp. A) is not known as only adults of each species were collected and in small numbers or as single specimens.

Undoubtedly there have been many unreported introductions of non-indigenous fishes that have subsequently not established. Assuming the number of known introductions represents only one half of all actual introductions, this latter number is still only a very small fraction of the potential pool of approximately 2000 species represented by all non-indigenous fish species currently maintained in aquaria or allowed to be imported into Australia. List 1 of the EPBC Act (2002) contains about 1500 prescribed fish species that can be imported into Australia. There are probably at least a further 500 non-approved species maintained in Australia that were present before establishment of import restrictions or subsequently imported illegally (R. McKay, *pers. comm.*). Interestingly, this estimate (25% of the total pool) is similar to the percentage of non-prescribed cichlid species (30%) of the total number recently offered for sale by wholesalers in NSW (Arthington *et al.* 1999). This proportion is even smaller if only aquarium species are considered.

While the total number of known species introduced into Australia is now at least 36, twenty-nine species are aquarium or ornamental species. Only three of these latter species can be clearly identified as non-accidental introductions: the Guppy and Mosquitofish that were deliberately introduced for bio-control, and the European carp for aquaculture and recreational fishing. The majority of the other eight, non-aquarium species (e.g., Trout, Salmon, Tench and Redfin perch) were introduced for recreational fishing by Acclimatisation Societies in the 19th century. Also, in terms of overall numbers of fish, this potential pool is probably dominated by a much smaller number of taxa. In 1999, ninety-two percent of the volume of the seven million fish imported fish into Australia were from only seven taxa: goldfish (Cyprinidae: 22%); tetras (Characidae: 19%); livebearers (Poeciliidae: 16%); barbs (Cyprinidae: 11%); cichlids (Cichlidae: 10%); catfish (Bagridae; Siluridae; Loricariidae; Heptapteridae; Pimelodiidae; Mochokidae; Callichtyidae; Gyrinocheilidae: 8%); and gouramis (Belontiidae: 6%) (Kahn *et al.* 1999). While the vast majority of ornamental fish imported (80%) are tropical varieties originating from ornamental fish farms in S.E. Asia and S. Asia, there is increasing domestic production of these fish due to increased import costs and shortages of supply. The six most popular imported species are the Goldfish, Neon tetra, *Paracheirodon innesi*, White cloud mountain minnow, Guppy, Platy and Swordtail (Kahn *et al.* 1999). Of these, only one species, the Neon tetra (the world's most popular aquarium species), has not been reported from open waters in Australia.

Several other popular aquarium fishes in Queensland, including tetras and other characins, many catfish species and barbs have been reported elsewhere in open waters (e.g., continental USA and Hawaii), but the great majority have not established breeding populations. In Australia, of these, only two barb species, *Puntius conchonius,* and *P. tetrazona,* have been reported from Queensland waters (Kailola *et al*. 1999). According to Allen *et al*. (2002), breeding populations of *P. conchonius* were present in streams in South Brisbane several years ago, but surveys in recent years have failed to detect these species (J. Johnson, Queensland Museum, Brisbane, *pers. comm*.). Undoubtedly, many of these fish have probably been discarded and failed to establish in open waters throughout Queensland and their presence gone unreported. Many of these fish, such as the tetras, catfish and barbs originate from acidic waters of the Amazonas in South America and may find local conditions unfavourable for long term survivorship, while some of these species, such as the tetras and barbs are relatively small and brightly coloured and also likely to be vulnerable to large, visual predators.

The simple indices used in the study as measures of propagule pressure and habitat matching did provide reasonably good indicators of introduction success – that is, establishment of breeding populations – although there were exceptions which reflect limitations in the methodology and data. At least four species had no range overlap (Mosquitofish, Jewel cichlid, Green terror and Burton's haplochromis). Two of these species (Green terror and Burton's haplochromis) had not previously been introduced into other countries outside their indigenous range, and both species were reported from only 1 or 2 sites respectively, with the Green terror reported only as a single specimen. Previous history of introductions is an imprecise measure of propagule pressure: a single introduction may fail if it involves a single individual or monosex individuals but may succeed if they are pregnant females (livebearers, such as the poeciliids) or incubating adults (mouthbrooders, such as the Mozambique mouthbrooder). There is very little detailed ecophysiological information for more than half the species reported in northern Queensland fresh waters, with little consistency in the measures used by different researchers to define physiological tolerance limits. Besides reproductive potential, reproductive mode is probably an important factor in establishment success. Of the eleven species that have established breeding populations, eight species have reproductive strategies which enhance survivorship of eggs and fry: live bearing (all poeciliids) (Baylis 1981; Vargas and De Sostoa 1996), mouthbrooding (Mozambique mouthbrooder and Burton's haplochromis) (Oppenheimer 1970; Philippart and Ruwet 1982; Subasinghe and Sommerville 1989) and surface bubble-nest building (the anabantoid Three-spot gourami) (Adler 1975; Lucas 1986; Jaroensutasinee and Jaroensutasinee 2001), by reducing exposure to adverse conditions such as low oxygen concentrations, variable water levels and presence of predators.

Absolute range overlap is also limited as an indicator of establishment success as it does not incorporate localised effects of climate, such as those due to altitude. Computer models based on climatic matching, such as BIOCLIM/ANUCLIM and CLIMEX have been used to predict the distribution of terrestrial flora and fauna, particularly in relation to climate change or spread of introduced pest species (e.g., Busby 1988; Boag *et al.* 1995; Sutherst *et al.* 1998; Samways *et al.* 1999; Duncan *et al.* 2001; Kriticos and Randall 2001). Such models incorporate latitudinal, altitudinal and climatic data based on temperature and rainfall and obviously provide a more precise means of mapping predicted distributions than simple range overlap, although such techniques have not yet been applied to non-indigenous fishes. While such models can provide better predictions compared with the relatively simple indices in this study, the precision of these models has been variable (see Samways *et al.* 1999; Duncan *et al.* 2001). For example, Boag *et al.* (1998) found that the CLIMEX model, using temperature and rainfall data, overestimated the areas of risk of establishment of a New Zealand flatworm, a major predator on earthworms, into new environments as it did not include soil acidity that limited earthworm distribution. Application of GIS techniques, incorporating soil pH and land use information, allowed a more detailed prediction of the flatworm's distribution and optimum areas of establishment and therefore the potential for damage to agricultural productivity.

Environmental factors other than temperature and rainfall have an influence on the survivorship and distribution patterns of freshwater fishes. These include salinity, pH, water hardness and oxygen concentration, as well as connectivity of habitat (i.e., absence of barriers to dispersal) and habitat integrity. Such models as CLIMEX may therefore overestimate areas of risk and likelihood of establishment, for example, for acidophile species such as the Green severum, as well as many barb, catfish and tetra species previously discussed. Other ecological attributes such as specialised reproductive strategy and trophic plasticity are important in the ability of non-indigenous fishes to adapt to, and survive in, new environments and need to be incorporated into these models. Opportunistic feeding is characteristic of many of the successfully established species, such as the poeciliids (e.g., Dussault and Kramer 1981, Pen *et al* 1993, Arthington and Marshall 1999) and some of the cichlids (e.g., Bowen and Allanson 1982; Janssens de Bisthoven 1990; Webb 1994) and the anabantoid Three-spot gourami (Rainboth 1996). Precocious reproductive maturity ('stunting') in cichlids such as the Mozambique mouthbrooder also provides the species with the ability to exploit a wide range of water bodies of differing depth and volume (de Silva and Amarasinghe 1989; Blühdorn *et al*. 1989), while mouthbrooding enhances survivorship of eggs and fry especially in habitats with unstable water levels and by providing protection from predators (Philippart and Ruwet 1982; Holden and Bruton 1994).

Further research to compare various predictive models applied to non-indigenous freshwater fishes is warranted. Clearly, more detailed ecological information is essential, particularly on physiological tolerances, not only of many prescribed species of non-indigenous freshwater fishes in Australia, but those species previously not imported and with no introduction history into open waters outside their indigenous ranges. Furthermore, in addition to a species' attributes as a potential invader and its invasion history, the biotic and abiotic milieu of the receiving waters needs to be considered in assessing the likelihood that such species will succeed or fail to establish. The following chapters investigated the interactions between non-indigenous fishes and resident predators and parasites, the use of refuges by non-indigenous fishes, and the possible role of habitat disturbance as a facilitator of the invasion process.

CHAPTER FOUR

OCCURRENCE OF NON-INDIGENOUS FISHES IN RELATION TO HABITAT AND PREDATORS: A CASE STUDY OF THE ROSS RIVER CATCHMENT

"All is fish that comes to the net"

Anon. 16th C proverb

4.1 Introduction

4.1.1 Habitats and invasions

According to community assembly theory, a community can be viewed as an open system susceptible to invasions from outside where invaders may be absorbed without loss of species, or where they may lead to extinctions (Drake 1991; Law and Morton 1996; Law 1999; Kooi *et al*. 1999; Kooi and Kooijman 2000). The community is therefore an assemblage of species constructed over time by a pool of potential invaders and each newly arriving species meets with what Elton (1958) called 'ecological resistance' (Post and Pimm 1983; Drake 1990). Moyle and Light (1996a) defined this resistance as three interacting elements: abiotic (or environmental), biotic and demographic. Moyle and Light suggested that virtually any species can invade and that any ecosystem can be invaded given the right circumstances, with successful establishment and integration depending on an interaction of all three elements. Different emphases have been placed on each element. Several studies (e.g., Li and Moyle 1981; Moyle *et al*. 1986, Case 1991 and Lodge 1993a,b) focussed on biotic resistance (mainly predation and competition) as most important, while others suggested that demographic factors, including life history traits such as high reproductive potential, reproductive strategy (e.g., mouthbrooding and ovovivipary), phenotypic plasticity (ability to shift from an altricial to precocial lifestyle in response to environment stress) and longevity had a significant role in increasing the probability of successful establishment (Arthington and Mitchell 1986; Bruton 1986; Lawton and Brown 1986; Crawley 1986,87; Rosecchi *et al*. 2001). In areas where abiotic conditions may be more harsh, environmental factors (e.g., hydrology, availability of cover) rather than community structure are usually more important determinants of invasion success or persistence of large populations of non-indigenous fishes (Pusey *et al.* 1993; Moyle and Light 1996b; Brown and Moyle 1997; Fausch *et al*. 2001), and stochastic events may also be important in determining invasion success (Gray 1986; Carlton 1996; Rejmanek 2000; Davis and Pelsor 2001).

Differences in invasion success can be related to refuges as protection against enemies, afforded by certain lifestyles (Hill and Lodge 1994; Hawkins and Gross 1992; Wonham *et al.* 2000). Many fish respond to predators by moving into protective habitats (Power *et al.* 1985; Rozas and Odum 1988; Savino and Stein 1989; Jordan *et al.* 1996a,b) and that the most vulnerable sizes (small adults, juveniles and fry) are found in the most protected habitats (Wilz 1971; Hall and Werner 1977; Ebeling and Laur 1985).

Chapman and Chapman (1996) and Chapman *et al*. (1996) found that wetland habitat associated with Lake Nabugado, Uganda, provided both structural and low-oxygen refuges for tolerant prey species from an introduced predator, the Nile perch, *Lates niloticus*. Nine out of 16 indigenous species, mainly cichlids, which disappeared from open-water habitat following the introduction of Nile perch, were found in these wetland habitats. They found a negative relationship among ecotones between species richness and dissolved oxygen, and a positive relationship between species richness and structural complexity. While they found it difficult to separate the effects of structure and low oxygen on the habitat use of Nile perch, they observed that the species was less abundant in wetland ecotones relative to exposed inshore areas, and were found even less frequently in wetland habitat with very low dissolved oxygen. They concluded that such habitats served as an effective refuge either because their structural complexity could reduce hunting efficiency, or because hypoxic conditions limited access by predators. Whether natural or as a result of anthropogenic habitat modification, such habitat can provide refuges for tolerant fishes that are potential prey for indigenous predators.

While habitat disturbance is not a prerequisite for invasion, non-indigenous species are frequently associated with disturbed environments (e.g., Arthington *et al.* 1983; Leidy and Fiedler 1985; Moyle 1986; Gehrke and Harris 2001). Moyle and Light (1996b) suggested that in aquatic systems with higher levels of human disturbance a much wider range of species can invade than in systems with low levels of human disturbance and that such invasions are consistent with the intermediate-disturbance hypothesis of Connell (1978). In the highly altered Sacramento-San Joaquin estuary, Moyle and Light reported that at least 30 exotic species were added to the indigenous fish community of 20 species, with only three subsequent extinctions (two indigenous and one exotic) which suggested that these exotic species may have an advantage over indigenous species more adapted to conditions prior to disturbance. It has also been observed that insular communities or depauperate communities are more prone to invasion. This may be due, in part, to lower biotic resistance of indigenous fauna that are poorly adapted to changed conditions associated with increased frequency of anthropogenic disturbance and where invaders are particularly effective competitors or predators (Elton 1958;

Holdgate 1960; Carlquist 1965; Sailer 1978; Moyle 1986; Vitousek *et al.* 1997). Williamson and Fitter (1996) reported that of the 70 species of birds introduced into Hawaii, over 70% occurred in modified habitat, particularly in the lowlands where virtually no indigenous habitat was left. Maciolek (1984) made similar observations of Hawaiian fish fauna where 76% of freshwater fish introductions have established. Maciolek concluded that attrition of indigenous fish fauna has largely been due to anthropogenic habitat modification that has favoured exotic species – virtually all of Hawaii's successful introductions have occurred on Oahu Island where 80% of the state's population occurs and habitat disturbance is the most severe. Moyle (1986) noted that wholesale replacement of indigenous fish fauna has occurred in many drainages to the west of the Rocky Mountains in North America which typically had highly endemic fish faunas with relatively few species. Most of these streams have been dammed, diverted and otherwise modified to create more lacustrine or constant flow regimes that favour the non-indigenous species that occupied such habitat in their native ranges. The introduced fishes seemed better adapted to the changed hydrological conditions and well equipped to eliminate the indigenous species through predation and competition coupled with higher reproductive success. Moyle and Light (1996b) also noted that depletion of many western indigenous fish populations and establishment of "more desirable" non-indigenous species was facilitated by the use of piscicides by US Fisheries managers to reduce any indigenous biotic resistance.

According to Bunn *et al*. (1997, 1998, 1999) dramatic declines in the health of coastal streams throughout Queensland is due to loss of riparian vegetation and proliferation of invasive introduced plants, such as ponded pasture grasses (e.g., Paragrass) and floating plants (e.g., Salvinia and Water hyacinth) as well as excessive growth of native aquatic vegetation in nutrient-enriched conditions. This proliferation has resulted in profound changes in channel morphology, loss of aquatic habitat, declines in water quality and changes in trophic structure of many local aquatic communities. Proliferation of aquatic vegetation, including invasive exotic species, can provide a refuge for tolerant indigenous or non-indigenous fish species. Arthington (1992) observed a strong habitat preference by introduced poeciliids for the edge of pools where paragrass had invaded urban streams in southeastern Queensland. A number of studies elsewhere have found that poeciliids, and other taxa such as cichlids that are a dominant component of the introduced fish fauna in Australia, show a strong association with vegetation, which has been interpreted as a response to predation (Bruton and Boltt 1979; Noltie and Johansen 1986; Winkelman and Aho 1993; Chick and McIvor 1997).

4.1.2 Ross River and non-indigenous fishes

The Ross River has experienced large-scale modification since the establishment of Townsville in the 1870s (see Chapter 3). Webb (1994) found that, prior to 1990 there was an assemblage of at least 21 indigenous fish species in the main channel of the weirs. Three non-indigenous species were also recorded in the catchment: the Mozambique mouthbrooder (a cichlid and a dominant component of the large fish assemblage in the weir main channel) and two poeciliids, the Platy and the Mosquitofish.

Nine of the indigenous species recorded from the weirs were piscivorous, mostly in low numbers, although some of the smaller species, such as the Mouth almighty (*Glossamia aprion*) and Spangled perch (*Leiopotherapon unicolor*), were common. Except for the Long-finned eel (*Anguilla reinhardti*), the large, catadromous species were under-represented in the weirs due to the presence of the barrages blocking upstream migration, although small numbers of some species were present either following translocation from below the first barrage (e.g., Tarpon and Trevally (*Charanx ignobilis*)) or limited stocking of cultured fish (Barramundi). In February 1992, as part of a Queensland Government recreational fishing enhancement program, annual mass stocking of Barramundi into the Ross River weirs began, with approximately 70,000 fingerlings stocked between 1992 and 1997. No downstream movement of mature fish occurred between 1991/92 and 1997/98 as no water flowed over the weir barrages due to the prolonged El Nino, which affected summer rains in northern Queensland (Chapter 3). The Barramundi population, therefore, increased substantially during this period. The most recent species stocked by QFS in the weirs, was the Mangrove jack, *Lutjanus argentimaculatus*, in 2002 after completion of this study.

An eight-month sampling program in Aplin and Black Weir was started in 1991 by Webb (1994) just prior to the start of mass stocking of Barramundi fingerlings in the river in early 1992, and this sampling program was repeated in 1992/93. In Australia, there have been no similar stocking programs in impoundments where large non-indigenous fish populations are present and pre-stocking catch data are available. While this precluded a replicated field study, the annual stocking of Barramundi in the Ross River weirs since 1992 provided an opportunity to assess the possible longer-term impacts on populations of non-indigenous fishes of mass stocking of a predator, and the relative importance of non-indigenous fishes as prey species for indigenous piscivores.

The objectives of this study were to:

- assess changes in fish assemblages in the Ross River by comparing the biomass and numbers of indigenous and non-indigenous freshwater fish species in gillnet catches from the Ross River weirs between Barramundi pre-stocking (1991/92) and post-stocking (1997/98) periods;
- identify impacts of Barramundi stocking on Mozambique mouthbrooder
- investigate the diets of common piscivores
- assess the relationship between habitat disturbance and the pattern of distribution, diversity and abundance of non-indigenous fishes in the Ross River catchment; and
- assess the relative importance of these fishes as prey species, particularly for medium-sized indigenous piscivores.

4.2 Methods

4.2.1 Gill net sampling program

A gillnet sampling program conducted in 1991/92 and 1992/93 in the Ross River weirs (Webb 1994) was repeated in 1997/98 using the same gillnets and standardised sampling methods. Samples were collected over the same 8-month period from four sites, two in Aplin Weir and two in Black Weir (see Figure 4.1 and Table 4.1 for sampling locations). Each site was sampled every two months. The gill nets (mesh size: 5 cm, 7.5 cm, 10 cm, 11.5 cm, 12.5 cm, 15 cm, 18 cm) were set from, and perpendicular to, the river bank about 1.5 hours before sunset and retrieved after three hours.

Fish retained for further laboratory analysis (including piscivorous species) were euthenased in an ice slurry, then later packed in ice and stored overnight in a cold room. In the laboratory, the total length (TL) and weight of each fish were recorded. Also, for Mozambique mouthbrooder males and females, the maximum body depth (BD) was recorded. Measurement details for piscivorous fish retained are included in Section 4.2.3. Fish that were not to be retained were measured in situ (total length and body weight) and then returned to the water.

Non-parametric statistical analyses were performed for Aplin Weir and Black Weir catch effort data for comparison of numbers and biomass of fish species caught per sampling day between 1990/91 and 1997/98, and for comparisons of sex-ratio, and morphometric data of Mozambique mouthbrooder populations in the weirs. The biomass and morphometric data provided information about the distribution of age (size) classes in the catches, and possible changes in these parameters not necessarily discernible from catch numbers.

4.2.2 Seine net sampling program, 1999-2000

Populations of small fish were sampled using a small seine net (3.5 m x 1.6 m; 5 mm mesh) over an eight-month period from September 1999 to April 2000. Samples were collected at sites in the Aplin Weir (Bush Garden and Palmetum), Black Weir (Loam Island and Apex), at two sites in the Ross Dam, and two sites in the Upper Ross River (see Figure 4.1; Table 4.1).

Samples were also collected from sites in the catchment isolated from the main channel of the Ross River. These sites were narrow channels between small islands and the main bank in Aplin Weir (Bush Garden site – Figure 4.2) and Black Weir (Loam Island site – Figure 4.3), a seasonally isolated wetland adjacent to the main channel of the Upper Ross River (Figure 4.4) and three small creeks which flow into Aplin Weir: Campus Creek, Regatta Creek (Figure 4.5) and Cranbrook Creek (Figure 4.6).

The narrow island channels were blocked at both ends by dense mats of vegetation, mainly introduced Water hyacinth and Paragrass, but also dense growth of submerged macrophytes, mainly the indigenous Hornwort (*Ceratophyllum demersum*), non-indigenous Fanwort and emergent Water lilies (*Nymphaea* spp. and *Nymphoides* spp.) and Lotus lily *(Nelumbo nucifera*). The junctions of the creeks with the main channel were similarly blocked by dense vegetation. The shallow, low-lying wetland was separated from the immediate floodplain of the Upper Ross River by a roadway. Emergent Swamp paperbark (*Melaleuca* spp.) were common on higher ground and the deeper pools contained dense beds of submerged macrophytes dominated by Hornwort, with emergent Water lilies also common. During extended dry periods, this swamp can become a series of pools or dry completely. It is effectively isolated by the roadway, except after heavy summer rains when water overflows the road from the wetland into the Upper Ross River via a series of old streambeds and sand extraction channels (Figure 4.4).

Pre-dawn dissolved oxygen readings were taken between September and December at sites blocked by aquatic vegetation. Readings were taken at approximately 10-20 cm below the surface in isolated pools, within the vegetation “barrier” and in the adjacent main channel of the Ross River. Median minimum dissolved oxygen levels in the vegetation and isolated pools were very low (0.5 and 1.7 mg/L respectively), and well below the concentration for the main channel (7.5 mg/L)(see Appendix C, Table C1 and Figure C1).

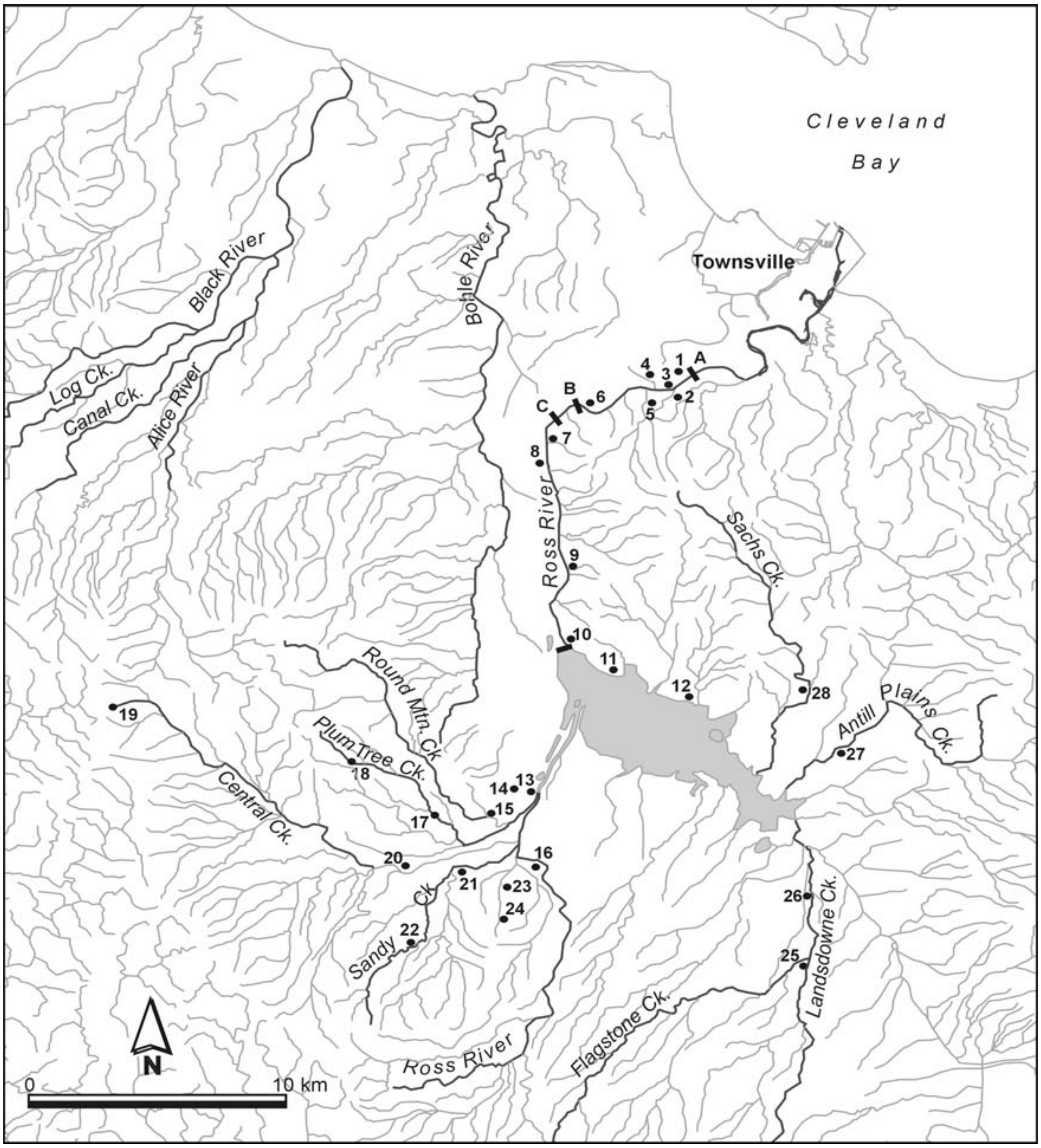


Figure 4.1 Fish sampling sites in the Ross River catchment, (refer to Table 4.1)

Table 4.1 Sampling site locations in the Ross River catchment (refer to Figure 4.1)

Site no.	Sample Method	Site location (for site coordinates, refer to Table B1, Appendix B)
1	Seine	Bush Garden, Aplin Weir
2	Seine	Campus Creek (lower, mid reach)
3	Gill	Palmetum, Aplin Weir (AW1)
4	Seine	Cranbrook Creek, Cranbrook
5	Seine	Regatta Creek, Riverside Gardens
6	Gill	Aplin Weir (AW2) below Gleeson Weir barrage
7	Gill	Black Weir (BW1), immediately upstream of barrage
8	Seine	Loam Island, Black Weir
9	Gill	Apex (BW2), approx. 5km upstream from BW1
10	Seine	Immediately below Ross Dam spillway
11	Seine	Ross Dam (S1); on Mt Stuart side of Dam
12	Seine	Ross Dam (S2), on Mt Stuart side of Dam, approx 2km from S1
13	Seine	Upper Ross main channel (S1); confluence of dam and old Ross River channel
14	Seine	Upper Ross Wetland, on Round Mountain side of Dam adjacent to Upper Ross (S1) main channel
15	Seine	Round Mountain Creek
16	Seine	Upper Ross main channel (S2) approx. 2km upstream from S1
17	Obs/Dip	Plum Tree Creek (mid reach)
18	Obs	Plum Tree Creek (upper waterhole)
19	Obs	Central Creek (upper reach – spring)
20	Obs	Central Creek (lower-mid reach)
21	Seine	Sandy Creek (lower-mid reach)
22	Seine	Sandy Creek (upper reach)
23	Seine	Un-named creek (lower reach)
24	Obs	Un-named creek (upper reach)
25	Seine	Landsdowne Creek (lower)
26	Seine	Landsdowne Creek (mid)
27	Seine	Antill Plains Creek
28	Seine	Sachs Creek (mid)

At each site samples were collected at least 5 m apart. Sampling areas were relatively shallow (<1.75 m) with a low gradient within about 3 m of the bank and where the aquatic vegetation was sufficiently patchy to allow dragging a seine net. In the blocked island channels, where open water was restricted to pools of varying size, samples were taken as far apart as possible. At each sampling site, samples were collected from five locations selected at random from 10 previously established locations. Standard net tows were done where one person remained close to the bank holding the net while a second person swept the other end of the net back toward the bank in an arc. At the bank edge the net was lifted like a scoop and the catch placed in a large aerated container. Fish were then identified to species and the number of individuals of each species less than about 5 cm in length were counted and placed in a second aerated container. Samples were then returned to the water, or kept temporarily in the container if sampling areas were less than about five metres apart. Larger piscivorous fishes caught were also recorded. Samples of the piscivorous Mouth almighty were collected using a small seine net (3.5 m x 1.6 m x 5 mm mesh) from the same sampling sites. Fish were euthenased in an ice slurry then packed in ice and taken to the laboratory for examination.

Figure 4.2 Bush Garden, Aplin Weir, Ross River, blocked channel with small island on the right

Figure 4.3 Loam Island channel in foreground (Note: vegetation blocking entrance to the Ross River in background)

Figure 4.4 Upper Ross isolated wetland site (Note: the Ross River Dam bund wall in the background)

Figure 4.5 Mouth of Regatta Creek, Aplin Weir, Ross River (Note: dense Water hyacinth in foreground and main river channel in background)

Figure 4.6 Mouth of Cranbrook Creek, Aplin Weir, Ross River (Note: creek heavily overgrown with Paragrass and main river channel in background**)**

Figure 4.7 Ross River Main Channel site, Aplin Weir, near mouth of Regatta Creek (Note: lack of emergent and floating marginal vegetation)

Comparisons of catch composition were made between restricted sites and their nearest main channel sites using the non-parametric Kruskal–Wallace test with the Bonferonni adjustment applied to the critical alpha level, set at 0.01 for non-normal data.

A fish survey of small streams in the Upper Ross River and Ross Dam catchment, where water was flowing or present as permanent waterholes, was also conducted in Sept 2000 (see Figure 4.1 and Table 4.1 for site locations). Fish were collected in the early morning after sunrise using a small seine net (as above), a 1 m x 1 m dip net, or observed directly with binoculars.

4.2.3 Gut contents of indigenous, piscivorous fishes in the Ross River catchment

Gill net samples (1997/98)

Fish were weighed (total wet weight) and measured (total length (TL) and body depth (BD). Gape- width (the internal width of the lower jaw with the mouth open) was measured using calipers. The gut contents of each fish were examined and food items identified and the number of each food item recorded. The total length and body depth of whole prey items (fish and crustaceans) were also measured. The total percentage composition of each food item for the gut content of each predator species was calculated. The composition of major prey components in gut samples of Barramundi collected from the Aplin Weir and Black Weir were compared using the non-parametric Kruskal-Wallace test with an alpha level at 0.01 for non-normal data.

Seine net samples (1999/2000): Mouth almighty

Each fish was measured (total length; body-depth; gape-width; total weight) and allocated to one of three size classes: Small (<7.00 cm); Medium (7.05 to 10.00 cm); and Large (>10.05 cm). The gut contents of each fish were examined and food items were identified and the number of each food item recorded. The total length and body-depth of whole prey items (fish and crustaceans) were also measured.

Between-site and location comparisons of gut content composition were done using the non-parametric Kruskal-Wallace test with the alpha level set at 0.01 for non-normal data. The percentage composition of the major component species (>1% of total content) in Mouth almighty gut samples was compared with the percentage of major component species in seine net samples above from the same sampling locations in the Ross River using the non-parametric Kruskal-Wallace test with the alpha level set at 0.01 for non-normal data.

4.3 Results

4.3.1 Catch data analysis

Species composition of gillnet samples

Fourteen indigenous and two non-indigenous fish species were collected in gillnets from the Aplin and Black Weirs between 1990 and 1998 (Table 4.2). In 1997/98, ten species were collected in samples from both weirs, which included the two non-indigenous cichlids, the Mozambique mouthbrooder and the Midas cichlid (the latter collected only in Aplin Weir). The indigenous Archerfish, not collected in 1991/92 and 1992/93, was a minor component of the catch in 1997/98. Six other indigenous species (Milkfish, Giant trevally, Hyrtl's tandan, Mouth almighty, Spangled perch, Long-finned eel and Tarpon), collected in 1991/92 or 1992/93, were not collected in 1997/98. The Milkfish and Giant Trevally were present only as minor components (≤ 2 individuals) of the total catch in the 1991/92 samples.

Table 4.2 List of fish species collected in gillnets and seine nets in samples from the , freshwater reaches of the Ross River catchment, 1990 to 1998. * translocated indigenous species; ** possible translocated indigenous species

Common name		Gill	Seine	Upper Ross
Indigenous fish				
Bony bream	*Nematolosa erebi*	3	3	3
Black catfish	*Neosilurus ater*	3	3	
Hyrtl's tandan	*Neosilurus hyrtlii*	3		3
Spangled perch	*Leiopotherapon unicolor*	3	3	3
Banded grunter	*Amniataba percoides*	3	3	3
Archerfish	*Toxotes chatareus*	3	3	
Freshwater longtom	*Strongylura krefftii*	3	3	
Tarpon**	*Megalops cyprinoides*	3		
Barramundi*	*Lates calcarifer*	3		
Sleepy cod*	*Oxyeleotris aporos*	3	3	3
Mouth almighty	*Glossamia aprion*	3	3	3
Long-finned eel	*Anguilla reinhardti*	3	3	
Milkfish**	*Chanos chanos*	3		
Giant trevally**	*Charanx ignobilis*	3		
Fly-specked hardyhead	*Craterocephalus st stercusmuscarum*		3	3
Fire-tailed gudgeon	*Hypseleotris gallii*		3	3
Empire gudgeon	*Hypseleotris compressa*		3	
Purple-spotted gudgeon	*Mogurnda adspersa*		3	3
Speckled goby	*Rediogobius bikolanus*		3	3
Eastern Qld rainbowfish	*Melanotaenia sp splendida*		3	3
Agassiz's glassperch	*Ambassis agassizi*		3	3
Non-indigenous fish				
Mozambique mouthbrooder	*Oreochromis mossambicus*	3	3	
Midas cichlid	*Amphilophus citrinellum*	3	3	
Jewel cichlid	*Hemichromis guttatus*		3	
Guppy	*Poeciliia reticulata*		3	
Mosquitofish	*Gambusia holbrooki*		3	3
Platy	*Xiphophorus maculatus*		3	
Swordtail	*Xiphophorus helleri*		3	

Six medium to large, non-indigenous cichlid species (Oscar, Green terror, Green severum and Convict cichlid), and two smaller species (Firemouth and Burton's haplochromis), recorded from below the Ross Dam (Chapter 3), were not collected in the gill nets.

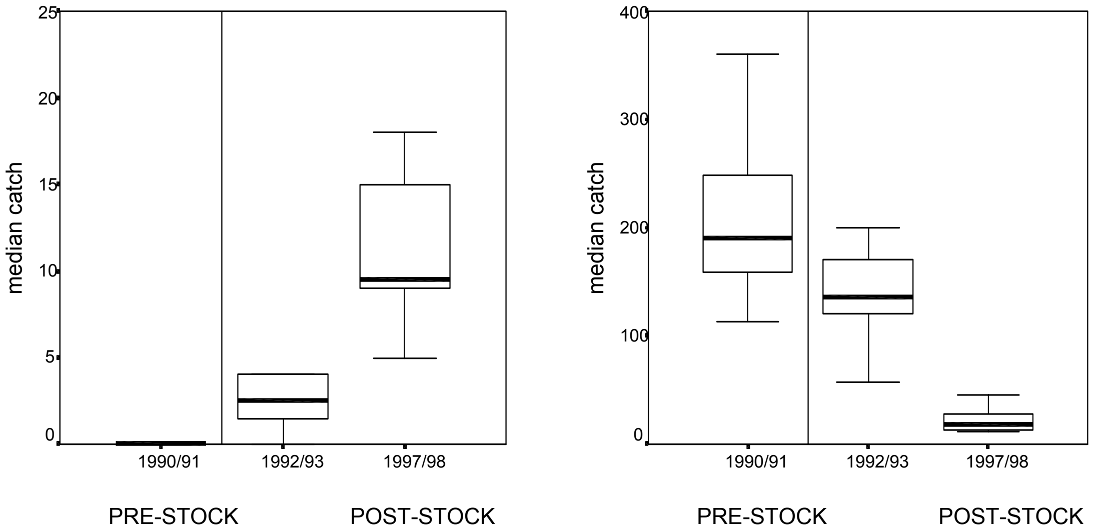
Between-years (1991/92 and 1997/8) comparison of catch

There were significant decreases in numbers and biomass of small to medium-sized indigenous fishes (Bony bream, Spangled perch) in Aplin and Black Weir, and significant decreases for the small indigenous Mouth almighty and Banded grunter and medium-sized Hyrtl's tandan in Black Weir between the Barramundi pre-stocking and post-stocking sampling periods (Table 4.3 and Figure 4.8). In 1997/98 there were no Hyrtl's tandan, Spangled perch and Mouth almighty collected, and only four Banded grunter collected from both weirs. In 1991/92, Bony bream was the dominant medium-sized indigenous fish in the weirs. In 1997 total catches for Bony bream in Aplin Weir and Black Weir decreased by 53.7% and 89.4% respectively in comparison with the 1991/92 total catches. For the large indigenous Black catfish and the non-indigenous Mozambique mouthbrooder there were no changes in numbers over the same period.

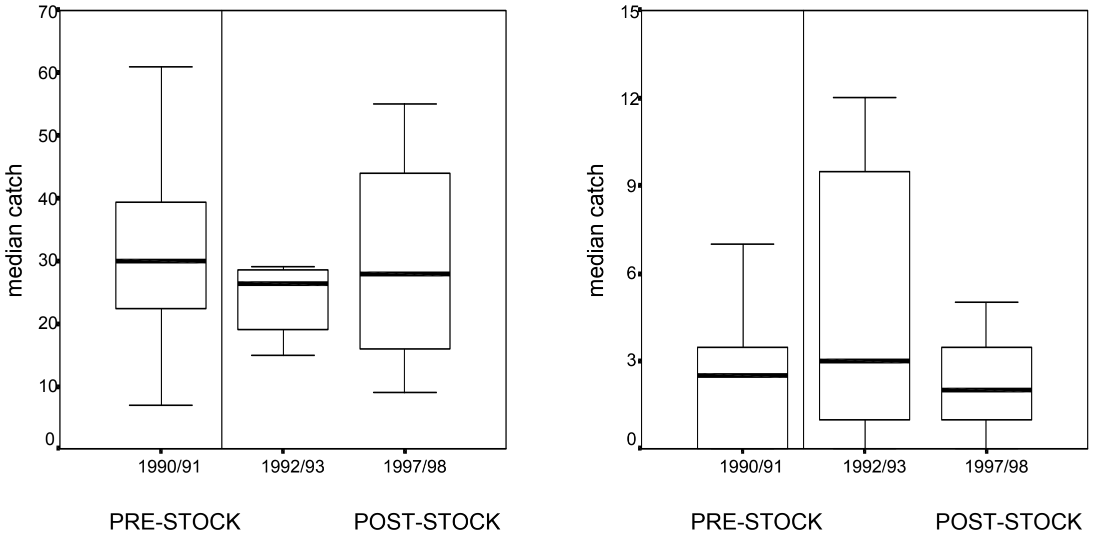
Table 4.3 Statistical summary for comparison of gillnet catch data (biomass and numbers) for Barramundi pre-stocking (1991/92) and post-stocking (1997/98) surveys In the Aplin Weir and Black Weir, Ross River (***Non-indigenous fish)**

	APLIN WEIR							
	BIOMASS (g) (median)				**NUMBERS (median)**			
Species	91/92	97/98	χ^2	p	91/92	97/98	χ^2	p
Bony bream	31105.7	14785.6	11.568	0.003	178	86	8.647	0.003
Hyrtl's tandan	79.0	0.0	5.974	0.049	0.5	0.0	4.885	0.027
Spangled perch	207.1	0.0	7.098	0.029	1.5	0.0	8.471	0.004
Banded grunter	19.4	0.0	8.443	0.015	1.5	0.0	5.128	0.024
Mouth almighty	0.0	0.0	1.117	0.572	0.0	0.0	1.000	0.317
Barramundi	0.0	18696.4	15.058	0.001	0.0	6.5	12.387	<0.001
Freshwater longtom	352.3	307.9	4.093	0.129	1.0	1.0	0.048	0.826
Sleepy cod	0.0	719.2	6.968	0.031	0.0	1.0	8.471	0.004
Tarpon	297.3	0.0	7.747	0.021	1.0	0.0	6.55	0.010
Black catfish	440.4	2533.8	6.038	0.049	0.5	3.5	3.264	0.071
Mozambique mouthbrooder*	14055.5	32694.9	6.471	0.039	31	35	0.099	0.753
Midas cichlid*	314.6	0.0	8.937	0.011	1.5	0.0	6.536	0.011

	BLACK WEIR							
	BIOMASS (g) (median)				**NUMBERS**			
Species	91/92	97/98	χ^2	p	91/92	97/98	χ^2	p
Bony bream	30873.4	2516.0	11.294	0.001	191.0	19.0	11.327	0.001
Hyrtl's tandan	207.85	0.0	4.873	0.027	2.0	0.0	6.35	0.010
Spangled perch	798.6	0.0	10.87	0.001	5.0	0.0	8.862	0.003
Banded grunter	297.3	0.0	7.085	0.008	5.5	0.0	6.841	0.009
Mouth almighty	149.6	0.0	8.388	0.004	1.5	0.0	8.471	0.004
Barramundi	0.0	19925.7	12.886	<0.001	0.0	9.5	12.973	<0.001
Freshwater longtom	1362.7	0.0	7.085	0.008	2	0.0	4.098	0.043
Sleepy cod	0.0	0.0	0.408	0.523	0.0	0.0	0.019	0.890
Black catfish	1991.0	1469.2	0.011	0.915	2.5	2.0	0.011	0.915
Mozambique mouthbrooder*	16747.3	27323.7	1.064	0.172	30	28	0.011	0.916

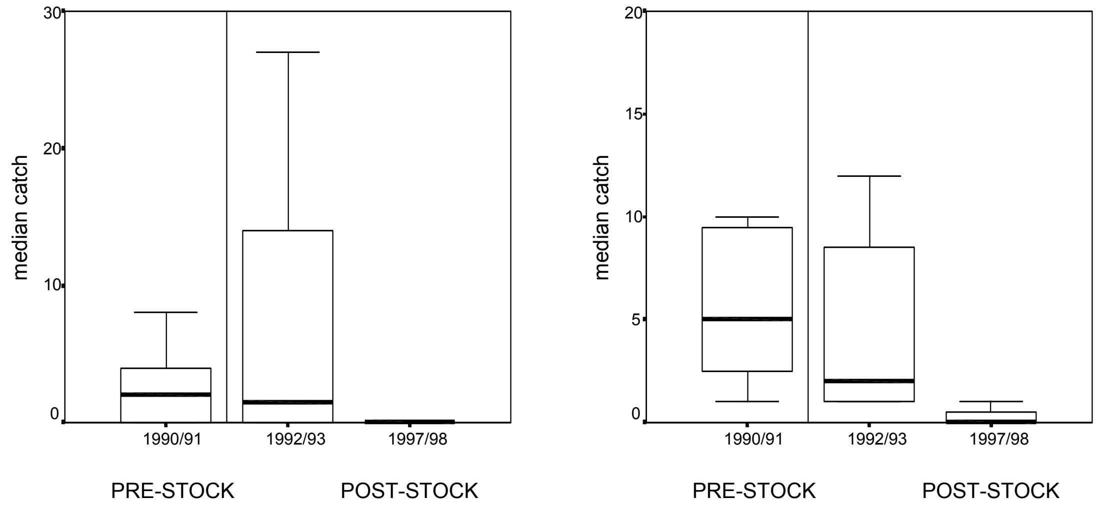

a. Barramundi

b. Bony bream


c. Mozambique mouthbrooder

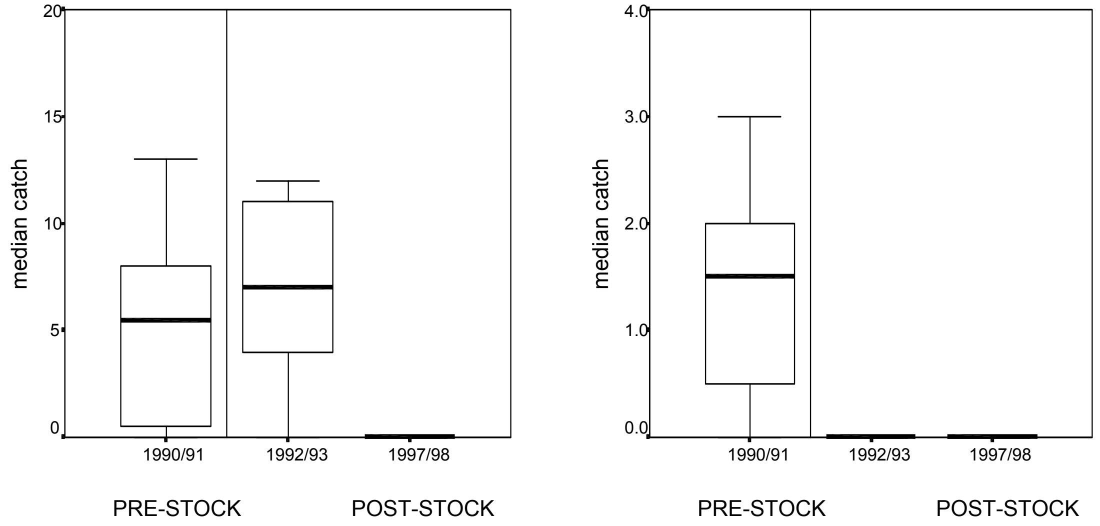
d. Black catfish

Figure 4.8 Change in gillnet catch of fish species between pre-stocking and post-stocking of Barramundi in the Ross River weirs: a, Barramundi; b, Bony bream; c, Mozambique mouthbrooder; d, Black catfish; e, Hyrtl's tandan; f, Spangled perch; g, Banded grunter; h, Mouth almighty continued



e. Hyrtl's tandan

f. Spangled perch



g. Banded grunter

h. Mouth almighty

Figure 4.8 continued

In 1991/92 this species was the second largest component of samples from the Black Weir, but was the dominant component in 1997/98. It was the dominant biomass component in catches in 1997/98 from both weirs partly due to the decline in catches of Bony bream. 1n 1997, total biomass catches of Mozambique mouthbrooder for Aplin Weir and Black Weir increased by 110% and 55.6% respectively, even though change in numbers of fish caught in both weirs was ≤10% of 1991/92 catches. Over the same period, catches of Bony bream decreased in Aplin Weir and Black Weir by 69.9% and 90.1% respectively compared with 1991/92 catches. Gillnet catches of the non-indigenous fish species (Midas cichlid) decreased from (18) in 1991/92, to zero in 1997/98.

There was a significant increase in Barramundi in gillnet catches between pre-stocking and post-stocking periods. In 1997/98, Barramundi was the dominant biomass component in Black Weir samples and the sub-dominant component (after Mozambique mouthbrooder) in Aplin Weir samples. Three other large piscivores were collected – Tarpon, Freshwater longtom and Sleepy cod – all of which were minor catch components. Tarpon were only caught in small numbers in 1991/92 and none were caught in 1997/98. There was no change in the numbers of Freshwater longtom, while there was a small, but significant, increase in numbers of Sleepy cod compared with 1991/92 when none of this species was caught.

Mozambique mouthbrooder catch 1991/92 and 1997/98 comparison

A statistical summary of the catch data for Mozambique mouthbrooder collected from the Ross River weirs in 1991/92 and 1997/98 is presented in Table 4.4.

Table 4.4 Statistical summary of gillnet catch data for Mozambique mouthbrooder collected from the Ross River weirs, 1991/92 and 1997/98 (TL = Total length in cm., Wt = Body weight in grams)

year	**sex**	**n**	**APLIN WEIR**		**n**	**BLACK WEIR**		**ANOVA (between weirs comparison)**			
			TL	Wt.		TL	Wt	Total Length		Weight	
								U	p	U	p
91/92	Female	110	28.2	486.7	105	27.5	448.0	5141.5	0.199	4983.0	0.102
	Male	123	30.8	657.9	126	31.1	658.6	7291.0	0.419	7567.5	0.748
	Juv.	59	18.9	130.1	29	19.6	146.7				
97/98	Female	123	32.9	787.1	121	32.6	704.9	6716.0	0.188	5432.0	<0.001
	Male	152	39.0	1185.9	100	34.5	893.3	4926.5	<0.001	4251.0	<0.001
	Juv.	11	18.4	112.0	5	18.0	119.0				

In 1991/92, there was no significant difference in size or weight of males and females in catches between weirs. In 1997/98, males were significantly larger and heavier in catches from Aplin Weir compared with Black Weir catches. Females in Aplin Weir catches were significantly heavier than those from Black Weir although there was no significant difference in size between weirs (Table 4.4).

There was no significant difference in the proportion of males and females in total catches from weirs between 1991/92 and 1997/98 (χ^2 =0.784, p = 0.206). There was a significantly smaller proportion of juveniles (3.1%) in the total catch from weirs in 1997/98 compared with the proportion in the catch (15.6%) in 1991/92. Mozambique mouthbrooder males and females were significantly smaller in catches from the Ross River weirs in 1991/92 compared to catches in 1997/98 (Table 4.5). The proportion of large to small adult male and female Mozambique mouthbrooder in the total catch decreased significantly between 1991/92 and 1997/98. In 1991/92, small adult males were 35.3% of the catch, but were only 3.2% of the catch in 1997/98. In 1991/92, small adult females were 72.1% of the catch, but were only 16.0% of the catch in 1997/98 (Table 4.5).

Table 4.5 Statistical summary for between-years comparisons of Mozambique mouthbrooder sex and size class for gillnet catches from the Ross River weirs, 1991/92 and 1997/98 (TL = median total length, cm)

Sex	Year	N	Size class		TL (cm)	Size comparison		Size class comparison	
			large	small		U	p	X^2	p
Male	91/92	249	161	88	31.0				
Male	97/98	252	244	8	37.3				
						10072.5	<0.001	82.678	<0.001
Female	91/92	215	60	155	28.3				
Female	97/98	243	204	39	32.7				
						7659.5	<0.001	146.743	<0.001

4.3.2 Gut content analyses of piscivorous fish in the Ross River catchment from gillnet samples (1997/98)

Barramundi

A summary of the gut contents of fish collected from sites in the Aplin Weir and Black Weir is presented in Table 4.6. A total of 522 prey items were recorded from the guts of 134 Barramundi and consisted of 15 taxa (10 fish, two crustacean, one turtle and two molluscan). Fish constituted 78.74% of the total gut contents, with 95% of the remainder consisting of the two crustacean taxa, *Cherax* sp. and *Macrobrachium* sp.. For samples from the Aplin Weir and Black Weir, the dominant components were Bony bream (*Nematolosa erebi*) (50.5 and 21.0% respectively), Fly-specked hardyhead *(Craterocephalus stercusmuscarum*) (19.7 and 19.0% respectively) and *Macrobrachium* sp.

Table 4.6 Percentage composition of gut content samples of Barramundi collected from the Ross River Weirs, 1997/98

	Aplin Weir			Black Weir		
Site	**Palmetum**	**Gleesons**	**ΣWeir**	**Black1**	**Apex**	**ΣWeir**
Prey	%	%	%	%	%	%
Fly-specked hardyhead	19.7	19.5	19.7	16.7	21.5	19.0
Bony bream	64.4	11.0	50.5	22.8	18.3	21.0
Banded grunter	0	0	0	0.9	0	0.5
Agassiz's glassperch	3.9	30.5	10.8	21.9	6.4	15.1
Spangled perch	0	1.2	0.3	0.9	10.7	5.4
Roman-nosed goby	0	0	0	0	1.1	0.5
Mouth almighty	3.0	4.9	3.5	12.3	1.1	7.3
Hyrtl's tandan	0.4	0	0.3	0	0	0
Fire-tailed gudgeon	0	1.22	0.32	0	0	0
Long-finned eel	0	0	0	0	1.1	0.5
Freshwater turtle	0.4	1.2	0.6	0	0	0
Cherax	0.4	0	0.3	1.7	0	1.0
Macrobrachium	7.3	30.5	13.3	20.2	39.8	29.3
Gastropod	0	0	0.3	0	0	0
Bivalve	0	0	0	2.6	0	1.5

sp. (13.3 and 29.3% respectively). Other important components were Agassiz's glassperch (*Ambassis agassizii*) (10.8 and 15.1% respectively) and Mouth almighty (12.3% for samples from Black 1 site, Black Weir). A total of 159 juvenile Bony bream were found in gut contents of a sub-sample of 12 Barramundi (size range: 52.1-74.7 cm TL) collected from the Aplin Weir in 1998, with a maximum of 25 recorded from one specimen (TL 61.5 cm). There were no non-indigenous fish species recorded from the gut samples of any piscivorous fish examined from the Ross River.

A summary of between-weir comparison of samples of main gut components is presented in Table 4.7. There was no difference in mean number for three of the five main components (Bony bream, Agassiz's glassperch and Mouth almighty) and a significant difference for the other two components (Fly-specked hardyhead and *Macrobrachium* sp.).

Table 4.7 Comparison of the main gut components between samples of Barramundi collected from the Aplin Weir and Black Weir, Ross River, 1997/98

Prey spp	**Bony bream**	**Agassiz's glassperch**	**Mouth almighty**	**Fly-specked hardyhead**	***Macrobrachium***
X^2	1.957	0.232	0.351	4.176	5.861
Degrees of freedom	1	1	1	1	1
p	0.162	0.630	0.554	0.041	0.015

Four other indigenous piscivorous fish species (Freshwater longtom (*Strongylura krefftii*), Sleepy cod (*Oxyeleotris lineolatus*), Spangled perch and Tarpon) were collected in gill nets from the Ross River weirs, although they represented minor components of the total catch. No non-indigenous fish were found in the gut contents of any of these species examined. Twenty-two Freshwater longtom were examined with only six specimens containing prey, comprising two fish species (83.3% Bony bream and 16.7% Banded grunter). Nineteen Sleepy cod were examined with only three specimens containing prey items, all *Macrobrachium* sp. Seven Spangled perch were examined with five specimens containing prey items, all of which were Fly-specked hardyhead. A total of 11 Tarpon were examined with no specimens containing prey items.

4.3.3. Seine net sampling program

Ross River weirs and adjacent creek and wetland site fish survey, 1999-2000

Seventeen indigenous species and seven non-indigenous species were collected in seine net samples from main channel and restricted sites in the Ross River catchment (Table 4.2). One indigenous species, the Purple-spotted gudgeon, was only collected at one restricted site (Upper Ross wetland) and not in main channel sites. Twelve indigenous species collected in main channel sites were also collected from restricted sites. Of the seven non-indigenous species, all were collected in restricted sites and only two of these species (Mosquitofish and Mozambique mouthbrooder) were collected in main channel sites.

In main channel sites, non-indigenous fish were a minor component of all catches (1.2% of total catch) (Figure 4.9), while the indigenous Fly-specked hardyhead was the dominant component with approximately 66% of the total catch. Bony bream juveniles were <1% of the total seine net catch from main channel sites although, as adult and sub-adult fish, they were a major component of gill net catches from the weirs (see section 4.2.3). Similarly, Mozambique mouthbrooder juveniles were <1% of the total seine net catch from main channel sites, while the adults and sub-adults were a major component in gillnet catches from the weirs (see Table 4.8 and 4.9).

In the samples from the two blocked island channels, indigenous fishes were the dominant component of the total catch (77.6%) (Figure 4.9). The dominant indigenous fish species were the Fire-tailed gudgeon (*Hypseleotris gallii*) (28.2%), Empire gudgeon (27.7%) and the Fly-specked hardyhead (17.2%). The non-indigenous Mosquitofish and the Platy made up 10.7% and 8.9% of the total catch respectively.

In the samples from the three blocked creeks (Figure 4.9) non-indigenous fishes were the dominant component of the total catch (73.3%). The dominant non-indigenous fish species were the Mozambique mouthbrooder (35.0%) and Mosquitofish (33.4%). The dominant indigenous fish species were the Fly-specked hardyhead (25.6%) and Empire gudgeon (20.0%). In the samples from the isolated wetland site in the Upper Ross catchment (Figure 4.9), the non-indigenous Mosquitofish was the dominant component of the total catch (61.5%). The dominant indigenous fish species was Agassiz's glassperch (30.7%).

Table 4.8 Summary statistics for seine net catches from Ross River weir sites, 1999/2000

	APLIN WEIR SITES						BLACK WEIR SITES					
site	Bush Garden			Palmetum			Loam Island			Apex		
spp	x	sd	%	x	sd	%	x	sd	%	x	sd	%
HH	181.0	291.59	86.1	110.2	65.53	76.6	74.6	53.20	51.9	37.3	47.27	57.1
FTG	114.5	9.40	5.5	9.5	9.50	6.6	40.1	35.61	27.9	2.6	3.38	4.0
EG	2.1	1.86	1.0	2.3	2.33	1.6	4.1	3.39	2.8	0.1	0.49	0.2
SG	6.4	10.99	3.0	7.2	9.49	5.0	14.9	11.99	10.4	-	-	-
MA	4.6	5.75	2.2	6.8	13.00	4.8	2.4	2.30	1.7	2.3	3.67	3.5
RF	1.1	2.25	0.5	3.7	5.15	2.6	2.5	2.42	1.8	8.9	15.54	13.6
OL	0.2	0.55	0.1	0.2	0.71	0.1	0.2	0.70	0.1	-	-	-
GP	0.1	0.31	0.1	0.6	1.57	0.4	1.9	2.43	1.3	10.9	7.61	16.8
BG	0.8	1.46	0.4	1.6	2.48	1.1	0.3	0.47	0.2	0.1	0.37	0.2
AR	-	-	-	0.1	0.16	0.1	-	-	-	-	-	-
TC	0.3	0.73	0.1	0.1	0.23	0.1	0.6	1.27	0.4	0.1	0.45	0.1
SK	0.3	0.80	0.1	0.1	0.23	0.1	-	-	-	0.6	1.19	0.9
BB	-	-	-	0.3	1.16	0.2	0.4	1.39	0.3	0.8	1.54	1.2
RNG	0.1	0.22	0.1	0.1	0.23	0.1	-	-	-	-	-	-
MM	-	-	-	0.1	0.23	0.1	-	-	-	1.0	2.31	1.6
GAM	1.6	4.82	0.8	1.2	2.75	0.8	1.65	1.98	1.1	0.4	1.27	0.6

Table 4.9 Summary statistics for seine net catches from Ross Dam and Upper Ross River sites, 1999/2000

	ROSS DAM						UPPER ROSS RIVER					
site	S1			S2			S1			S2		
spp	x	sd	%	x	sd	%	x	sd	%	x	sd	%
HH	42.0	43.50	69.0	56.5	44.3	70.5	37.0	29.86	40.8	50.8	33.02	51.3
FTG	6.0	6.07	9.9	10.3	9.1	12.9	18.3	14.81	21.2	19.2	19.84	19.4
EG	1.3	2.03	2.2	2.0	1.8	2.5	4.1	3.05	4.6	4.3	3.53	4.3
SG	1.6	2.94	2.7	3.1	3.9	3.9	0.7	2.77	0.8	0.1	0.37	0.1
MA	6.8	6.11	11.2	4.2	4.2	5.2	7.4	11.97	8.1	8.1	14.32	8.2
RF	0.6	1.39	1.0	0.4	1.1	0.5	3.6	3.76	4.0	4.9	4.62	5.0
OL	0.2	0.55	0.4	-	-	-	0.1	0.30	0.1	0.1	0.31	0.1
GP	0.6	0.88	1.1	1.9	4.0	2.4	12.3	25.91	13.6	3.8	8.79	3.8
BG	1.3	1.93	2.2	1.5	2.4	1.9	0.6	1.56	0.7	1.0	2.10	1.0
TC	-	-	-	0.1	0.7	0.2	0.2	0.54	0.3	0.4	0.60	0.4
SK	-	-	-	-	-	-	0.1	0.22	0.1	0.1	0.22	0.1
BB	-	-	-	-	-	-	3.4	16.58	3.7	5.5	23.42	5.6
SP	-	-	-	-	-	-	0.1	0.40	0.1	-	-	-
GAM	0.1	0.31	0.2	-	-	-	2.6	6.02	2.8	0.5	1.82	0.5

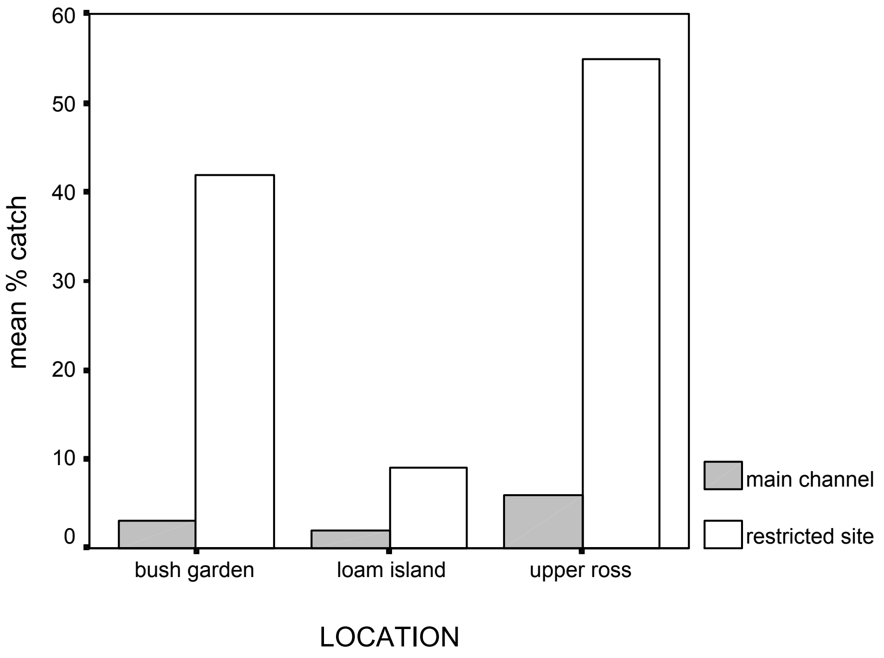
Indigenous species: HH = Fly-specked Hardyhead; FTG = Fire-tailed gudgeon; EG = Empire gudgeon; SG = Speckled goby; MA = Mouth almighty; RF = Eastern Queensland rainbowfish; OL = Sleepy cod; GP = Agassiz's glassperch; BG = Banded grunter; AR = Long-finned eel; TC = Archerfish; SK = Freshwater longtom; BB=Bony bream; SP=Spangled perch; RNG= Roman-nosed goby
Non-indigenous species: OM = Mozambique mouthbrooder; GAM = Mosquitofish

There were significantly larger numbers and more species of non-indigenous fishes in seine net samples from all six restricted sites compared with main Ross River channel sites (Table 4.10). There were no significant differences in numbers of indigenous fishes in either blocked island channel seine net samples, or the upper Ross wetland seine net samples, compared with main Ross River channel seine net samples (Table 4.10).

Table 4.10 Statistical summary for comparison of seine net catches between restricted sites and main channel sites in the Ross River catchment, for non-indigenous fishes (all species), indigenous fishes (all species) and indigenous piscivorous fishes, 1999/2000 (catch (IN) = median catch inside restricted site; catch (OUT) = median catch in main Ross River channel site)

FISH		**Bush Garden**	**Loam Island**	**Upper Ross Wetland**	**Regatta Creek**	**Cranbrook Creek**	**Campus Creek**
Non-indigenous	Sample size	20	20	20	20	20	10
	catch (IN)	39	10	139	60.5	56.5	10.5
	catch (OUT)	0	1	2	0	0	0
	χ^2	29.983	21.652	28.330	30.913	31.064	12.361
	p	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
	spp (IN) (median, max)	3 (4)	2 (3)	1 (1)	3 (3)	4 (5)	0.9 (3)
	spp (OUT) (median, max)	1 (1)	1 (1)	1 (1)	0 (1)	0 (1)	0 (1)
	χ^2	6.40	6.41	0.000	34.084	32.162	13.310
	p	0.011	0.011	1.000	<0.001	<0.001	<0.001
Indigenous	Sample size	20	20	20	20	20	10
	catch (IN)	63.5	119.5	84	21	1.5	25
	catch (OUT)	87.5	142	60	89	89	89
	χ^2	3.189	0.237	1.762	19.575	29.529	9.092
	p	0.074	0.626	0.184	<0.001	<0.001	0.003
	spp (median, max)	4.5 (7)	5.5 (12)	6.5 (7)	3 (6)	1 (2)	4 (5)
	spp (median, max) (OUT)	8.5 (12)	8.5 (11)	7 (13)	6 (8)	6 (8)	6 (8)
	χ^2	4.802	3.073	0.346	18.311	29.879	10.068
	p	0.028	0.080	0.557	<0.001	<0.001	0.002
Indigenous piscivores	Sample size	20	20	20	20	20	10
	catch (IN)	0	1	3.5	0	0	0
	catch (OUT)	4	4	6.5	8	8	8
	χ^2	12.092	15.151	5.338	19.116	26.518	33.473
	p	0.001	<0.001	0.021	<0.001	<0.001	<0.001

a. Blocked channel and wetland sites



b. Blocked creek sites

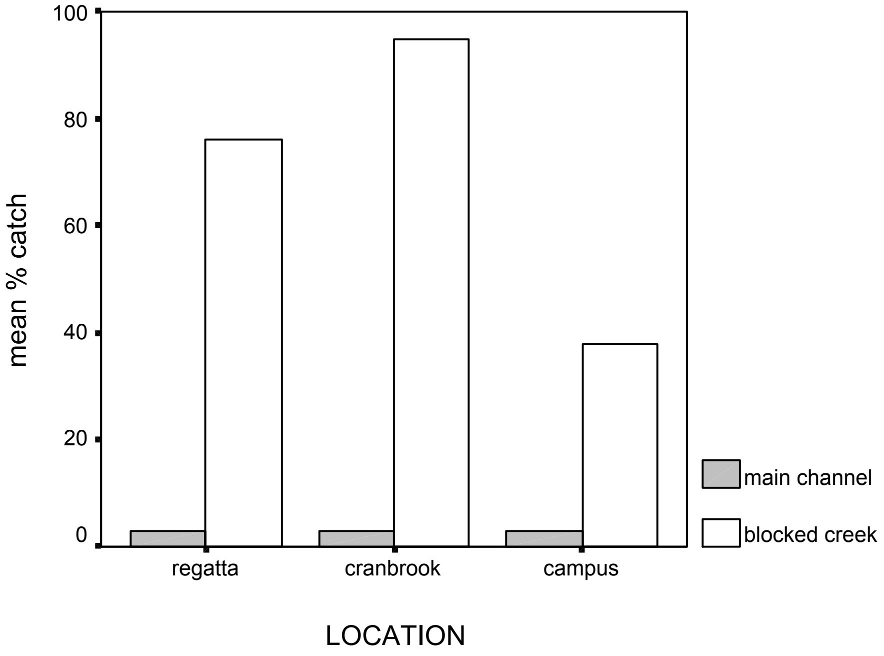
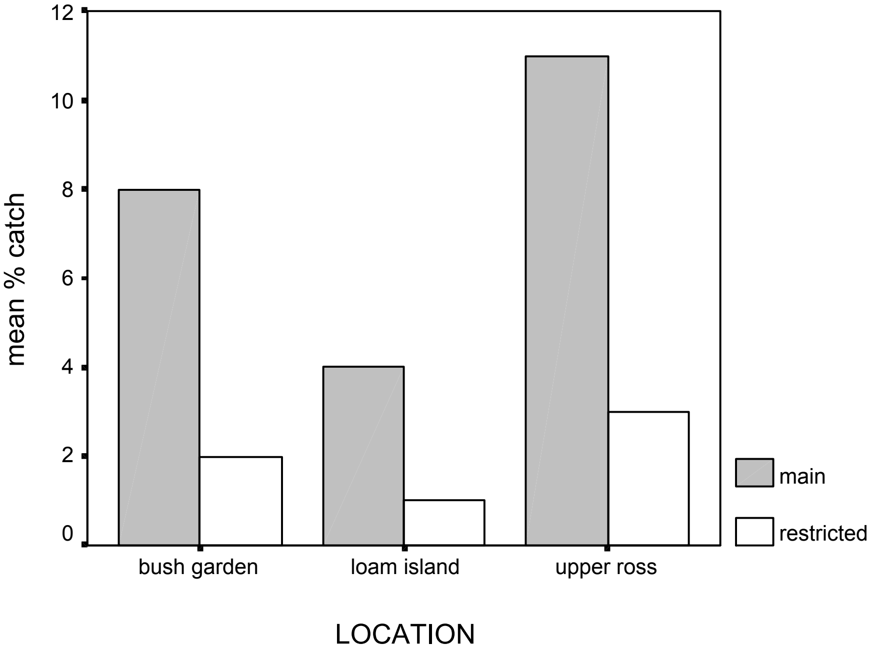


Figure 4.9 Percentage of non-indigenous fish in seine net samples from the Ross River catchment in comparison with main channel sites: a, from blocked island channel sites (Bush Garden and Loam Island) and upper Ross isolated wetland; b, blocked creek sites

a. Blocked island channel and wetland sites



b. Blocked creek sites

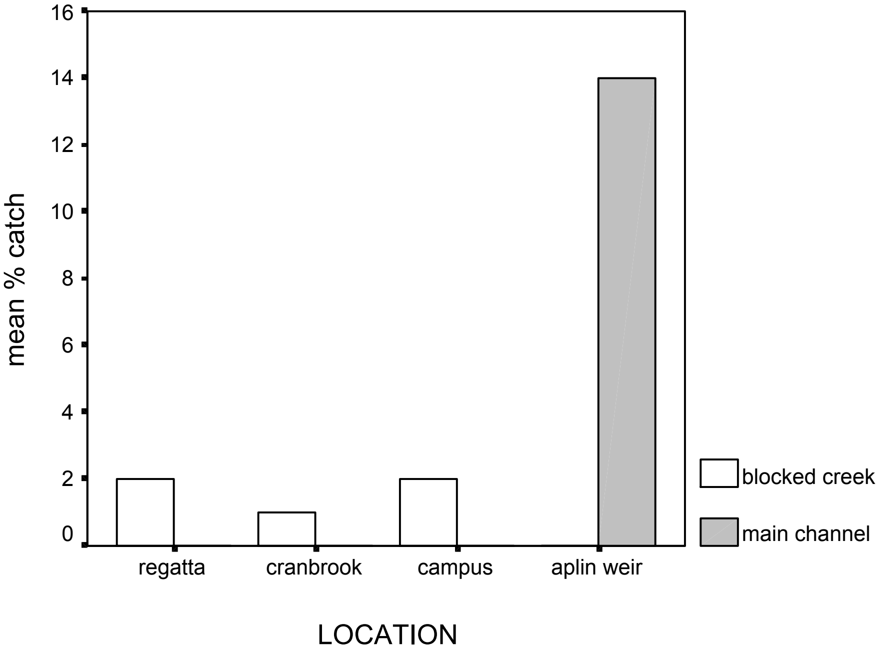


Figure 4.10 Percentage of indigenous piscivorous fish in seine net samples compared with main channel sites in the Ross River catchment: a, blocked channel sites (Bush Garden and Loam Island) and upper Ross isolated wetland; b, blocked creek sites

There were significantly fewer species of indigenous fishes in samples from blocked island channels compared with samples from main Ross River channel sites, although there was no significant difference in species number for the upper Ross wetland site samples compared with main Ross River channel samples (Table 4.9). There were significantly fewer individuals and species of indigenous fishes in samples from all blocked creek sites compared with samples from Ross River main channel sites (Table 4.10).

There were significantly fewer indigenous piscivorous fishes in all restricted site samples compared with main Ross River channel samples (Table 4.10 and Figure 4.10). Adults of medium-sized indigenous piscivores (Mouth almighty, Spangled perch, Purple-spotted gudgeon (*Mogurnda adspersa*)) and juveniles of larger indigenous piscivorous fish (Long-finned eel, Sleepy cod, Freshwater longtom) were minor components in virtually all samples collected from main channel sites, with an average combined composition of 7.6% of the total sample catch. These fish were minor components in all restricted site samples with an average combined composition of 0.9%. Very few piscivores larger than 5 cm were collected from restricted sites with none collected from Cranbrook Creek, and only a small number consisting of Mouth almighty and Sleepy cod juveniles collected from Regatta Creek and Campus Creek.

Upper Ross River and Ross Dam catchment fish survey, Sept 2000

Eight streams were surveyed which had either flowing water or permanent waterholes present (Figure 4.1). Twelve indigenous species and one non-indigenous species, the Mosquitofish, were recorded in these streams (Table 4.2). Some streams have been subject to disturbance, such as stock watering, localised sand extraction (Landsdowne Creek, Central Creek) and introduced plants, such as Salvinia and Water hyacinth (lower reaches of Plum Tree Creek and Round Mountain Creek) and Fanwort (lower reach of Sachs Creek). However, they were in a less disturbed condition than sites surveyed in the suburban reaches of the catchment: the riparian vegetation was intact in the middle to upper reaches and, where water was present, there was very little instream macrophyte growth. In Landsdowne Creek, while the water was turbid, there were scattered patches of macrophytes in the shallow margins and large numbers of indigenous fish present, including Eastern Queensland rainbowfish (*Melanotaenia splendida splendida*), Fire-tailed gudgeon, Agassiz's glassperch and piscivorous Spangled perch. No non-indigenous fish were recorded at this site. In the two streams where Mosquitofish were found (Sandy Creek and Central Creek), no predatory fish were recorded and very little instream vegetation was present.

4.3.4 Gut content analysis Mouth almighty

A total of 424 prey items were recorded from 254 Mouth almighty collected from main channel sites in the Ross River weirs, Ross Dam and Upper Ross River (Table 4.11). There were 11 prey taxa, consisting of six fish species (Fly-specked hardyhead, Mosquitofish, Banded grunter, Fire-tailed gudgeon. juvenile Mouth almighty and Speckled goby), one amphibian tadpole, two crustacean taxa (*Macrobrachium* sp. and an atyid sp.) and two aquatic insect taxa (Odonata and Ephemoptera). For the total gut contents, fish were the dominant component (68.6%), with Fly-specked hardyhead (52.4%) the most frequently recorded.

Table 4.11 Summary statistics for total prey items from gut samples of Mouth almighty collected from main channel sites in the Ross River weirs, Ross Dam and Upper Ross River, 1999/2000

Prey taxa	n	median	mean	se	max	min	%
Fly-specked hardyhead	222	1.0	0.880	0.075	8	0	52.4
Fire-tailed gudgeon	49	0.0	0.190	0.038	5	0	11.6
Banded grunter	1	0.0	0.004	0.039	1	0	0.2
Mouth almighty	17	0.0	0.069	0.016	1	0	4.0
Speckled goby	1	0.0	0.004	0.039	1	0	0.2
Mosquitofish	1	0.0	0.004	0.039	1	0	0.2
Atyid	35	0.0	0.140	0.026	3	0	8.2
Macrobrachium	78	0.0	0.310	0.039	4	0	18.4
Odonata	13	0.0	0.051	0.014	1	0	3.1
Gyrinid	1	0.0	0.004	0.039	1	0	0.2
Dipteran larva	1	0.0	0.004	0.039	1	0	0.2
Ephemoptera	3	0.0	0.012	0.068	1	0	0.7
Tadpole	2	0.0	0.008	0.005	1	0	0.5

Other important components were the two crustacean taxa and the Fire-tailed gudgeon while Mouth almighty, dragonfly larvae (Odonata) and the remaining seven taxa combined (2.4%) were present as minor components. The only non-indigenous fish species recorded was the Mosquitofish found in samples collected from one site (Bush Garden) in Aplin Weir. The Fly-specked hardyhead was the dominant component for all locations, comprising 35.4 to 71.5% of the total prey composition.

There was a significantly higher number of Fly-specked hardyhead in samples from Aplin Weir compared with Black Weir. Hardyheads were 71.5% of total gut components from Aplin Weir samples and 38.8 % of components from Black Weir samples. There were no differences in composition of samples between sites above the Dam spillway (Ross Dam and Upper Ross) and no difference between locations (above and below the dam spillway) (Table 4.12).

On all but three occasions occasions, total catch numbers of Mouth almighty were less than five specimens. For the September 1999 sample from the Loam Island site, gut contents consisted of two fish species (Fire-tailed gudgeon, 25.0% and Fly-specked hardyhead, 12.5%), two crustacean species (*Macrobrachium* sp., 25.0% and an atyid sp., 25.0%) and one odonate species (12.5%). For the January 2000 sample from the same site, gut contents consisted of three fish species (Fly-specked hardyhead, 6.7% Fire-tailed gudgeon, 73.3% Mouth almighty fry, 13.3%) and one crustacean species (atyid sp., 6.7%). For the March 2000 sample from the Bush Garden site, Aplin weir, gut contents consisted of one fish species (Fire-tailed gudgeon, 40.0%) and one crustacean species (*Macrobrachium* sp., 60.0%).

Table 4.12 Statistical summary for between sites and between locations comparisons of prey taxa in gut samples of Mouth almighty collected from the Ross River weirs, Ross Dam and Upper Ross River, 1999/2000

	sites		**sites**		**locations**	
	Aplin-Black Weir		**Ross Dam- Upper Ross**		**Above-Below Spillway**	
locations	U	p	U	p	U	p
Fly-specked hardyhead	1783.5	0.001	1470.0	0.654	7097.0	0.118
Fire-tailed gudgeon	2257.0	0.057	1476.0	0.542	7722.0	0.527
Mouth almighty	2482.5	0.375	1518.0	0.793	7737.0	0.427
Speckled goby	2520.0	1.000	1510.5	0.304	7865.0	0.256
Atyidae	2416.5	0.453	1429.0	0.228	7879.5	0.859
Macrobrachium	2090.0	0.021	1431.0	0.394	7925.0	0.979
Tadpole	2457.0	0.208	1539.0	1.000	7825.5	0.212
Gyrinidae	2488.5	0.375	1539.0	1.000	7881.0	0.378
Odonata	2311.5	0.023	1534.5	0.946	7896.0	0.855
Diptera	2480.0	0.260	1539.0	1.000	7881.0	0.378
Ephemoptera	2520.0	1.000	1453.5	0.073	7722.0	0.048

Comparisons of the relative percentage of fish species in the gut contents and seine net samples of small fishes collected from the Ross River weirs, Ross Dam and Upper Ross River are presented in Table 4.13. For all seine net samples and gut content samples, Fly-specked hardyheads were the dominant component and Fire-tailed gudgeons the sub-dominant component. For Aplin Weir, there were no differences for six out of seven comparisons of fish species in Mouth almighty gut samples and in seine net samples. There was significantly lower percentage of Speckled goby (*Rediogobius bikolanus*) (0%) in the Mouth almighty diet compared with seine net samples (3.6%). For Black Weir samples, there were no differences for six out of seven comparisons. Eastern Queensland rainbowfish was a significantly smaller percentage in the Mouth almighty diet (0%) compared with seine net samples. For Upper Ross samples, there were no differences for five out of seven species percentage comparisons. Eastern Queensland rainbowfish and Agassiz's glassperch were smaller percentages of the

Table 4.13 Comparison of percentage composition of small fish in seine net samples and in gut samples of Mouth almighty collected from the Ross River weirs, Ross Dam and Upper Ross River, 1999/2000

	APLIN WEIR				BLACK WEIR				ROSS DAM				UPPER ROSS			
spp	% env	% ma	χ^2	p	% env	% ma	χ^2	p	% env	% ma	. χ^2	p	% env	% ma	χ^2	p
HH	83.2	89.5	0.579	0.447	53.5	60.0	5.147	0.025	69.9	63.7	0.622	0.430	40.8	68.5	6.408	0.011
FTG	5.6	4.8	0.667	0.414	20.4	36.0	0.05	0.824	11.6	23.1	0.000	0.989	20.2	25.9	0.221	0.638
SG	3.6	0.0	10.35	0.001	7.1	0.0	5.248	0.022	3.4	1.9	3.309	0.069	-	-	-	-
MA	3.0	4.0	0.326	0.568	2.2	4.0	0.256	0.618	7.8	7.7	0.358	0.550	8.1	5.5	0.734	0.392
RF	1.2	0.0	7.045	0.008	5.5	0.0	8.059	0.005	-	-	-	-	4.0	0.0	11.63	0.001
GP	-	-	-	-	6.1	0.0	11.632	0.001*	1.8	0.0	3.379	0.066	13.6	0.0	8.06	0.005
BG	0.6	0.8	0.953	0.329	-	-	-	-	2.0	0.0	4.454	0.035	-	-	-	-
BB	-	-	-	-	-	-	-	-	-	-	-	-	3.7	0.0	1.860	0.173
GAM*	0.7	0.8	0.319	0.572	1.0	0.0	4.080	0.043	0.1	0.0	0.409	0.523	2.8	0.0	6.110	0.013

HH = Fly-specked hardyhead; FTG = Fire-tailed gudgeon; SG = Speckled goby; MA = Mouth almighty; RF = Eastern Queensland rainbowfish; GP = Agassiz's glassperch; BG = Banded grunter; BB = Bony bream; GAM = Mosquitofish*

* Non-indigenous species

Mouth almighty diet (0% and 0%) compared with seine net samples (4.0% and 13.6% respectively). There were no differences between percentage of fish species in the Mouth almighty diet and seine net samples from the Ross River Dam.

4.4 Discussion

4.4.1 Gillnet catch data and large-bodied, non-indigenous fishes in the Ross River

The Ross River catchment, like many waterways in northern Queensland, is a highly modified system and impacted by various disturbances including construction of dams and barrages, high nutrient inputs, loss of riparian vegetation and gravel extraction. There is still is a diverse indigenous fish population within the freshwater reaches although catadromous species are absent, except for the Long-finned eel and the translocated Barramundi, Sleepy cod and, since 2002, the Mangrove jack, *Lutjanus argentimaculatus*. However, there is now a large assemblage of non-indigenous species present. The study showed that two factors – mass stocking of a predator (Barramundi) and habitat disturbance, in particular excessive growth of marginal aquatic vegetation – have had important influences on the indigenous fish community and on the establishment and distribution of non-indigenous fishes within the catchment. There is little evidence from gill net catch data and gut content analyses to demonstrate that indigenous piscivores, in particular, Barramundi, have affected population sizes of Mozambique mouthbrooder in the weirs directly by predation, but the presence of these predators might have increased the use of refuges by juvenile and sub-adult fish. The lack of piscivorous fishes and the high abundance and diversity of small, non-indigenous fishes, including Mozambique mouthbrooder juveniles, in areas isolated from the main river channel by dense vegetation suggest that these habitats provide a refuge from predation for these fishes (Lowe-McConnell 1982; Pusey *et al.* 1993; Chapman and Chapman 1996).

Gravel extraction and dredging activity can have a significant impact on aquatic communities and may have been a contributory factor in catch declines in Black Weir between 1991/92 and 1997/98. Physical disturbance can result in loss of habitat including spawning or nursery sites (Chutter 1969; Cote *et al*. 1999; Jutila *et al*. 1999; Lepage *et al.* 2000), reduced photosynthetic production due to increased turbidity (Havens and James 1991; Chessman and Williams 1999; Lewis *et al*. 2001), demersal smothering and clogging of fish gills (Davies-Colley and Smith 2001) and reduced feeding rates of fish (Rowe and Dean 1998;), although, in some instances, survivorship may increase due to reduced predation pressure by visual predators (Johnson and Hines 1999; Chigbu 2000; Bonner and Wilde 2002). Jackson (2000) reported that catches of the channel catfish, *Ictalurus punctatus*, in upstream reaches of the Yalobusha River, Mississippi, not subject to dredging were twice that of catches in downstream reaches where dredging occurred.

The data suggest that the major cause in decline of catches, particularly of Bony bream, was due to increased predation pressure rather than dredging activity. There was a negative association between the gillnet catches of several indigenous fish species and numbers of Barramundi in both Aplin Weir and Black Weir following the start of mass stocking. Numbers of the dominant Bony bream, in particular, decreased dramatically between the pre- and post-stocking sampling periods. Bony bream is a turbidity-tolerant species like other ileophagous fishes, such as gizzard shads (Ross *et al*. 1993), and has planktonic eggs unlikely to be affected by suspended sediments. Prior to and during the study, most large-scale dredging activity occurred in Black Weir for gravel extraction, and some localised pipeline construction and bridgework. Monthly turbidity levels tended to be higher than those for Aplin Weir, although both water bodies, while seasonally isolated, showed intra- and inter-seasonal changes in turbidity associated with changes in rainfall patterns, which are typical of tropical coastal streams (Chapter 2 – Section 2.3.5) and probably within the tolerance levels of most local indigenous fishes. Removal of aquatic vegetation by dredging, however, especially in the vicinity of the Apex sampling site in Black Weir could have adversely affected local fish populations, for example, by loss of cover and food.

Bony bream is an important prey species for many indigenous piscivores (Briggs and McDowall 1996) and has also been stocked in several Queensland impoundments as principal fodder fish for translocated Barramundi (Herbert and Peeters 1995). In 1997/98, Bony bream juveniles were the dominant fish component (50.5 and 21.0%) in gut content samples of Barramundi from both weirs and also in samples of Freshwater longtom. Population densities of Barramundi in the weirs, and therefore predation pressure on prey populations, annually increased after initial stocking, as mature fish were unable to migrate downstream due to lack of water flow over the barrages between 1992 and 1998. Significant impacts on Bony bream populations in the weirs are indicated by the marked decrease in sub-adult fish in gill net catches and the predominance of small juveniles, in some cases in quite large numbers (maximum: 25), in gut contents of relatively large Barramundi (50-100 cm TL). In contrast, no adults or juveniles of large, non-indigenous fish were found in the gut contents of any medium to large-sized piscivorous fish sampled in the weirs.

There was no difference in gillnet catches of non-indigenous Mozambique mouthbrooder between 1991/92 and 1997/98, but the species markedly increased as a proportion of catches due, in part, to the significant – and dramatic – decline in Bony bream catches in both weirs. For the Mozambique mouthbrooder, the dominant non-indigenous species in both weirs, there was a significant change in population size-frequency distribution rather than change in abundance in the period following the mass stocking of Barramundi. In 1990/91, smaller-sized Mozambique mouthbrooder (<22 cm TL) were 25.3% and 11.1% of the gill net catch of this species from the main channel sites in Aplin weir and Black weir respectively, while in 1997/98, the proportion of juveniles had decreased to 3.8% and 2.7% from these respective locations. The reduction in smaller-size classes may, in part, reflect increased predation, although the species was able to maintain recruitment to the adult population even with the significantly enlarged piscivore population in the weirs.

The change in Mozambique mouthbrooder catch composition may have been due to a shift in behaviour of smaller fish by increased use of refuges such as dense littoral vegetation. This is supported by observations of Mozambique mouthbrooder schools in the weirs: mixed-size class schools were commonly encountered prior to 1991, but in 1997, only schools of large-(adult) fish were observed; mixed-size populations of non-stunted fish, however, were observed in large ponds and in pools of creeks in the catchment (e.g., Campus Creek) where there were very few, or no, large piscivores present (unpublished data). These observations are consistent with studies which have shown that cichlid distribution in the open waters of main river systems and dams in eastern Africa is influenced by the presence of large piscivores (Jackson 1960; Donnelly 1969; Lowe-McConnell 1982). Where large piscivores were present, tilapia populations consisted of larger fish (greater than about 17 cm TL) which were able to avoid predation by gape-limited predators, while smaller juveniles remained in shallow water among dense vegetation. Where large piscivores were absent, tilapias were observed in mixed size classes of adults and juveniles at various depths.

Predation on small Mozambique mouthbrooder may also have been limited or reduced by reduction in the number of intermediate predators, such as the Mouth almighty. Webb (1994) found that Mouth almighty were a major component of the diet of Barramundi in the weirs after the start of the mass stocking in 1992, but they were a minor component of Barramundi gut samples collected in 1997/98. Mouth almighty is a relatively sedentary, ambush predator and therefore unlikely to be well represented in gillnet catches. However, heavy predation by Barramundi might have been, in part, responsible, for the absence in gillnet catches of the species in 1997/98. The comparison of prey proportions in gut contents of Mouth almighty with proportions of these fishes in seine net catches indicated that Mouth almighty fed on prey which were locally most abundant, that is, Fly-specked hardyheads and Fire-tailed gudgeons. Only one specimen of the non-indigenous Mosquitofish was found in gut content samples and none of any other non-indigenous species were found in this study, or in earlier studies, of Mouth almighty feeding in the weirs (Webb 1994,1995), although this may simply reflect the low occurrence of these fish in the main channel of the weirs.

4.4.2 Seine net data: small indigenous and non-indigenous fishes in refuge sites in the Ross River catchment

Unfortunately, due most probably to sub-optimal conditions, very few predatory fish, including Mouth almighty, were collected from restricted sites where non-indigenous fish were most abundant, so no detailed dietary comparison could be made with samples from main channel sites. The limited samples collected from the island channel sites, however, showed that Mouth almighty were feeding on the most abundant species present: the Fly-specked hardyhead and Fire-tailed gudgeon. Both of these indigenous species are fusiform in shape and soft-finned. The latter species, in open water, is relatively slow swimming in comparison with the more mobile Fly-specked hardyhead, although small juveniles of the species are much slower swimmers than adults. Significantly lower representation in gut samples of deeper-bodied, spinous species, such as the indigenous Eastern Queensland rainbowfish and Agassiz's glassperch, may also indicate a preference by Mouth almighty for more fusiform or slower-swimming prey (see Chapter 5). The absence of Mozambique mouthbrooder juveniles from gut samples of Mouth almighty may be due to predator avoidance or the relative inaccessibility (or low density) of these fish even in the main channel of the weirs. However, Mouth almighty have been observed in shallow exposed margins of a regulated stream on the Atherton Tablelands, northern Queensland, feeding on vulnerable, newly-released fry of the Mozambique mouthbrooder (E. Collins, *pers. comm.*). Small to medium-sized predators, such as Mouth almighty, could have an important limiting effect on recruitment in Mozambique mouthbrooder populations by predation on early juvenile stages where there is limited access to effective refuges, such as dense vegetation (see Chapter 5).

The seine net catch data showed that there was an association between habitat disturbance and the distribution of non-indigenous fish species within the Ross River catchment. Mosquitofish was the only non-indigenous fish found above the Ross Dam, in the Dam itself and in the old Ross River channel, particularly in the lower reaches adjacent to the wetland site. The species was found in the middle or upper reaches of two out of eight small streams surveyed which flow into the Upper Ross River or the Ross Dam. These upper catchment streams, although used for stock watering and sand extraction, were less disturbed than streams in the lower reaches of the Ross River. The number of non-indigenous species increased from the upper to the lower reaches of the catchment and both species number and abundance increased significantly within the “restricted” sites compared with main channel sites. This trend of increasing numbers of non-indigenous fish from headwaters to lower reaches of streams in urban and industrial areas has been observed elsewhere (e.g; Tramer and Rogers 1973; Moyle and Nichols 1973). Arthington *et al*. (1983) found a strong association between the level of disturbance in suburban Brisbane creeks in southeastern Queensland and occurrence of non-indigenous fish species. None of these species were found in undisturbed upper reaches but their number and abundance increased while those of indigenous species decreased in the lower reaches. These trends coincided with modification to stream channels, including dam construction, and increasing invasion of exotic Paragrass into the channels. In the most altered parts of Kedron Brook and Enogerra Creek, Arthington *et al*. 1983 recorded similarly high abundances of non-indigenous species with combined abundances of the Mosquitofish and the Green Swordtail between 78 and 99% of the total population of fish.

In the Ross catchment, non-indigenous species were only 1.2% of the seine net catch in main channel sites compared with 49.8% of the catch in the ‘restricted’ (refuge) sites. In September 2003, further survey work at four of these sites (Bush Garden, Loam Island, Upper Ross Wetland, Regatta Creek, and Cranbrook Creek) and an additional isolated lagoon site adjacent to Ross Dam produced similar results: while indigenous fishes were the dominant component in the blocked island channel sites and non-indigenous fishes were the dominant component of catches in blocked creek sites, non-indigenous fishes were significantly higher in abundance and number of species compared with main channel sites where they were a minor component of total catches (<1%) (Appendix C, Figure C2 and C3) (unpublished data). Non-indigenous fishes were the dominant component of the catch in five of the six restricted sites sampled. At least seven of the non-indigenous fish species recorded in the catchment, all of them small to medium-sized species, were found only in the highly modified creeks and island channels blocked by dense vegetation. Except for Mosquitofish, all of the introduced species found in the main channel of the weirs were large-bodied, such as the Mozambique mouthbrooder, the adults of which are relatively free from predation by gape-limited predators. Of these species, the juveniles of at least the Mozambique mouthbrooder were found predominantly in disturbed (restricted) sites. The number of indigenous species and abundances were also much lower in these restricted sites compared with the main river channel.

Pusey *et al*. (1993) noted that preference for dense vegetation by many non-indigenous species, such as Mosquitofish, and low availability of the habitat type and other forms of cover against predators, could explain the near absence of these fish in downstream sections of the Mary River in southern Queensland. Webb (1997) observed differences in the distribution of the Mozambique mouthbrooder between disturbed and undisturbed sections of a rainforest stream on the Atherton Tablelands, northern Queensland. Mozambique mouthbrooder were absent from sites with intact riparian vegetation, little in-stream vegetation, and good flow through riffles and deeper, clear pools containing predators such as Spangled perch. Mozambique mouthbrooder was found only where riparian vegetation had been removed and Paragrass had overgrown and reduced water velocity along the banks, creating patches of still water habitat suitable for such sedentary species. The dense vegetation provided a refuge from visual predators, a rich source of food in the form of organic sediment carried downstream and trapped by the vegetation, or created *in situ* by decomposing plant material. The low water velocity conditions also provided ideal habitat for courtship and spawning behaviour of the species.

Overall, there were fewer indigenous species, including piscivores, and lower abundances in most restricted sites compared with the main river channel. These restricted sites represent sub-optimal conditions for indigenous fishes as demonstrated by their very low minimum dissolved oxygen levels (Appendix C, Table C1 and Figure C1), and the dense vegetation act as an additional physical and visual barrier to piscivorous fishes searching for prey. The indigenous Fly-specked hardyhead, Eastern Queensland rainbowfish, Agassiz's glassperch and the Empire gudgeon can tolerate quite low DO levels: at 1.91-2.90mg/l^{-1} (25-35% saturation, but are unable to tolerate prolonged (five days) exposure to very low DO levels <0.85mg/l^{-1} (10% saturation) with 100% mortality of all species (Pearson *et al*. 2003). In contrast, non-indigenous poeciliids and cichlids are typically able to tolerate extreme hypoxic conditions by branchial respiration at the narrow, oxygen-rich, air-water interface (Lewis 1970; Kramer and Mehegan 1981; Kramer and McLure 1981; Chervinski 1982) and by decreased respiratory and metabolic rates (Cech *et al*. 1985; Bayster 1996 in McKinsey and Chapman 1998; Weyl *et al*. 1998). *Gambusia holbrooki* can tolerate DO concentrations as low as 0.20mg l^{-1} (McKinsey and Chapman 1998), while the Mozambique mouthbrooder has survived in water with DO concentrations as low as 0.10mg l^{-1} (Maruyama 1958). These tolerance levels are lower than the minimum dissolved oxygen concentrations lethal to indigenous fishes that were recorded in the dense patches of aquatic vegetation blocking the channels and creeks in the Ross catchment. Dissolved oxygen levels in the shallower and less shaded restricted pools can, during periods of very high air temperature, decrease well below lethal levels for indigenous fishes and might account for their absence in some sites (e.g., Cranbrook, Campus and Regatta Creeks) (see Appendix C, Table C1).

Both Tarpon and Barramundi have been found to be effective predators in culture ponds of non-indigenous fishes including tilapia juveniles and small cyprinodontids elsewhere in these species' introduced ranges (Popper and Lichatowich 1975; Fortes (1979) *in* Guerrero 1982; Bedawi 1985; Rao and Ghosh 1986; Genodepa 1987; El Gamal 1992). The predators were stocked in the ponds to reduce competition for food with primary culture species and reduce recruitment (in tilapias) to prevent economically undesirable stunting of fish. As there was little or no cover in these ponds, the predators were very effective in reducing population numbers of the target species. In freshwater lagoons in the lower Burdekin region, predators such as Tarpon and Mouth almighty were found to be effective in reducing large populations of Mosquitofish after dense mats of floating water hyacinth were removed (Perna 2003). Pearson *et al.* (2003) found juvenile Barramundi (80-95 mm TL), were able to tolerate DO levels of 1.3 mg/l^{-1} at 29°C for 48 hrs, but levels below this were lethal. Interestingly, Tarpon can survive in hypoxic conditions as they are capable of air breathing and have a respiratory gas bladder that enables them to survive in anoxic conditions, but only if they have access to the surface (Burggren 1979; Heisler 1993; Geiger *et al.* 2000). Dense mats of vegetation therefore can act as an effective physical and/or physiological (hypoxic) barrier to indigenous predators and as a refuge from predation for non-indigenous fishes tolerant of low oxygen conditions.

In conclusion, except for one specimen of Mosquitofish, no non-indigenous fish were found in the gut contents of any piscivorous fish sampled in the Ross River. This finding is consistent with previous research in the same locations conducted since 1991/92 (Webb 1994). Gill net data and gut content analyses suggested that predation pressure was the likely cause of the decline in catches of several medium-sized indigenous fishes, particularly Bony bream, rather than habitat disturbance (gravel extraction), although disturbance may have been a contributory factor in at least one of the weirs. Increased predator populations in the weirs, however, had no effect on the population size of the large-bodied, non-indigenous Mozambique mouthbrooder, but may have affected population size frequency distribution of the species with smaller juveniles delaying entry into adult schools in open water by making increased use of refuges. In the Ross River catchment, small non-indigenous fishes, including juveniles of larger species, were found predominantly in those sites where access by larger predators was restricted by dense vegetation which might act as a physical or physiological (hypoxic) barrier, or by other physical barriers (e.g., shallow water, sand bars or causeways). However, they were absent from, or in very low numbers, in more open bank margin sites in the main river channel where access by large predators is less restricted, at least where vegetation was patchy and could be sampled with seine nets. The only non-indigenous species present in large numbers in the main channel of the weirs was the large-bodied Mozambique mouthbrooder. The adults of this species are effectively free from predation by gape-limited piscivorous fishes, including large Barramundi.

This case study provides clear evidence that anthropogenically disturbed habitat favours the establishment and persistence of non-indigenous fish species. Such disturbance, therefore, can have a key role in the invasion process by altering ecosystem dynamics that might provide an opening for opportunistic invaders capable of taking advantage of the changed conditions. Even in the presence of large piscivores in the main channel of the Ross River weirs, increased structural complexity of habitat, created by excessive growth of aquatic vegetation in shallower margins and blocking creek entrances, provided a very effective refuge, especially for fishes with wide ecological tolerances. Such refuges may therefore favour the persistence of some hardier non-indigenous species, and those more vulnerable to predation in open waters, such as slow swimming, fusiform poeciliids or juveniles of large-bodied fish such as the tilapiine cichlids.

CHAPTER FIVE

SURVIVORSHIP OF NON-INDIGENOUS FISHES IN RELATION TO PREDATION AND REFUGES

"No species can long maintain itself anywhere which cannot, in some way, find a sufficient supply of food, and also protect itself against its enemies"

S.A. Forbes (1880)

5.1 Introduction

The establishment and persistence of a non-indigenous fish species in a new environment will be determined by the species' interactions, with its abiotic environment (e.g., physiological tolerances to temperature, salinity, pH, dissolved oxygen), and other species as competitors (for food and space), predators or prey. Wide ecological tolerances and trophic generalism will enhance the potential of the newly arrived species to establish in a wide range of local conditions. Maintenance of these populations will be influenced by factors such as the species' reproductive potential and survivorship of juvenile stages to maturity, which will also, in turn, reflect the effectiveness of predator avoidance strategies, or even the absence of predation pressure. According to Murdoch and Bence (1987), both indigenous and introduced predators in freshwater systems have frequently been recorded as driving local populations of their prey extinct, particularly in closed systems such as lakes and ponds (e.g., Hurlbert and Mulla 1981; Ogari and Dadzie 1988; Pont and Guillot 1991; Rowe 1993; Chapleau *et al*. 1997; Ogutu-Ohwayo 1999), but also in open stream systems (Englund 1999; McIntosh 2000). The outcome of predator-prey interactions can therefore depend on the ability of prey species to take advantage of refuges that afford protection against predation. Crawley (1992) stated that a "prey refuge exists when the predators are unable to drive prey to extinction" (p. 74). The refuge may reflect attributes of the prey, predator or the environment and be permanent or transient. Permanent refuges are those that physically exclude predators or predation while transient refuges do not exclude a predator or predation but increase the probability that the prey will elude predators in time or space. A prey refuge may not only be a physical "safe place", but can also be a "statistical" entity in the sense that some prey may find themselves in predator-free places by chance alone (Chesson 1978,1985; Pacala and Crawley 1992).

Prey-mediated refuges can include morphology such as body size or presence of spines that act as a deterrent, particularly in relation to gape-limited predators, which swallow their prey whole. Prey with body depths larger than the gape width of the predator effectively cease to be vulnerable to gape-limited predators (Broenmark and Weisner 1996; Persson *et al.* 1996; Dewey *et al.* 1997; Lundvall *et al.* 1999; Straile and Haelbich 2000; Nilsson and Bronmark 2000). The probability of survivorship of smaller prey in relation to the size of the predator may also be increased as a number of studies have demonstrated that gape-limited predators typically select prey of a size (i.e., body-depth) much smaller than the theoretical maximum (e.g., Lawrence 1958; Werner 1977; Gillen *et al.* 1981; Tonn and Paszkowski 1986; Hambright 1991). Schooling behaviour may provide "safety in numbers" so that the risk of predation for an individual may be reduced while increasing the likelihood of escape by confusing a predator with choice (Ross and Backman 1992; Magurran and Seghers 1994; Seghers and Magurran 1995; Barber and Huntingford 1996; Dommenici and Batty 1997; Nottestad 1998). Schooling may also decrease predator strike rate per individual within the group by dilution effect or group uniformity (Ranta *et al.* 1992; Watt *et al.* 1997; Peuhkuri *et al.* 1997; Peuhkuri 1999; Hoare *et al.* 2000). Individuals may minimise predation risk through speed and/or manoeuvrability (Chovanec 1992; Christensen 1996; Utne-Palm 2000; Chivers *et al.* 2001), by migrating away from predators (e.g., diel vertical migration (Vijverborg 1991; Dini and Carpenter 1992; Sydanoja *et al.* 1995; De Meester *et al.* 1995; Eckmann and Imbrock 1996), or changing activity periods to when predators are inactive (e.g., timing of invertebrate stream drift) (Poff *et al.* 1991; Flecker 1992).

Predator-mediated prey refuges may exist as a result of predator inhibition from attacking prey due to risk of predation by higher order predators (Godin 1986; Jakobsen *et al.* 1994; Utne *et al.* 1997) and cannibalism by larger con-specifics (Harvey 1991; Nilsson 2000), or predators switching prey with changes in prey availability (Hughes and Croy 1993; Rincon and Lobon-Cervia 1999; Wilhelm *et al.* 1999; Willette *et al.* 1999; Ohizumi *et al.* 2000), or ontogenetic dietary shifts (Olson 1996; Warburton *et al.* 1998). A number of studies have examined morphological differences and anti-predatory behaviour patterns among prey in the context of optimal foraging theory (e.g., Wahl and Stein 1988; Nilsson 2000). Schoener (1971) stated that organisms maximise their fitness by adopting feeding strategies that optimise the net energy yield per feeding time. He described optimal diets, foraging space, foraging time and foraging-group size in terms of a cost-benefit function, where pursuit and handling and eating times are an energy cost. Predators will select food that will provide a greater biomass yield per unit time (Le Brasseur 1969) and greater calorific value (Rozin and Mayer 1961), and minimise searching and handling time (Werner 1974; Kislalioglu and Gibson 1976a; Nilsson and Bronmark 2000). Consequently, gape-limited predators may select slower, smaller, fusiform or deep-bodied and soft-finned prey rather than more evasive, deep-bodied, spiny prey or size classes of the latter that are much smaller than their gape width (Werner 1974; Wahl and Stein 1988; Hambright 1991; Christensen 1996).

Foraging efficiency may also differ between predators with differing forage modes (e.g., ambush and patrol foraging) that can affect prey species survivorship (Eklov and Diehl 1994; Auster *et al*. 1995; Flynn and Ritz 1999). Eklov and Diehl (1994) observed that the Northern pike, an ambush predator, was more efficient around prey refuges (vegetation patches) which pike use for concealment, whereas the Redfin perch, an active predator, was more efficient in open areas such as the pelagic zones of lakes. Toline and Baker (1993) also found that differences in morphology (body depth) of northern Redbelly dace populations were correlated with differences in their foraging mode and prey species targeted. Populations of deeper-bodied fish with optimal morphology for acceleration and manoeuvering relied on ambush predation and caught higher numbers of evasive fish at high densities. In contrast, populations of shallower-bodied fish with optimal morphology for extended cruising relied on more active, wide-range foraging and caught less evasive food at low densities.

Habitat-mediated prey refuges may exist as a result of structural or physical features which prevent or limit access by predators and reduce foraging efficiency, e.g., shallow water (Harvey *et al*. 1988; Post *et al*. 1998; Englund 1999; Paterson and Whitfield 2001), waterfalls (Seghers 1974; Tate 1997), burrows, holes and crevices (Blake and Hart 1993; Gotceitas and Brown 1993; Nemtzov 1994; Caley and St John 1996; Holohan *et al*. 1998; McIntosh 2000), low water temperatures (Dale *et al*. 1999) and low O_2 concentrations (Chapman *et al*. 1995,1996; Malinen *et al*. 2001). Refuges may result from habitat heterogeneity which reduces encounters of visual predators with prey, including the presence of vegetation (Persson 1993; Eklov and Diehl 1994; Persson and Eklov 1995; Eklov and Persson 1995; Buckel and Stoner 2001; Anderson 2001), woody debris (Everett and Ruiz 1993; Crook and Robertson 1999); high turbidity (Abrahams and Kattenfeld 1997; Johnson and Hines 1999) and low light intensity (Goddard and Mathis 1997; McCartt *et al*. 1997), or reduction in detection by predators, for example, by the selection by prey of background or substrate to match their body shape, colouration or markings, often in combination with immobility (Maglio and Rosen 1969**;** Hautman and Dill 1994; Ramachandran *et al*. 1996; Gregory and Anderson 1997).

Efficient use of refuges can vary between different prey species (Schramm and Zale 1985; Savino and Stein 1982 1989), predator types or forage mode (James and Heck 1994; Burks *et al*. 2001) as well as differences in structural complexity of the refuge. Burks *et al*. (2001) observed that juvenile Redfin perch, *Perca fluviatilis*, were better at foraging for *Daphnia* within vegetation than Roach, *Rutilus rutilus*, which preferred to forage in more open water. Other studies (e.g., Persson and Eklov 1995; Jordan *et al.* 1996a,b; Savino and Stein 1982, 1989) also showed that the effectiveness of a refuge may depend on the degree of habitat complexity. Increased survivorship of prey was due to increased macrophyte density reducing the foraging efficiency of the predator. Persson (1993) also noted that although juvenile Roach and Redfin perch preferred open water, in the presence of predatory adult Redfin perch and Northern pike, both prey used vegetation as a refuge, but roach were more efficient than perch in evading predators in the vegetation.

Studies of introduced fishes have focused largely on their ecological tolerances (e.g., Castleberry and Cech 1986; Krupp 1992; Batty and Lim 1999) in relation to survivorship, or their trophic interactions where the species functions as a predator rather than as prey (e.g., Lounibos *et al*. 1992; Morgan and Buttemer 1996; Webb and Joss 1997; McIntosh 2000; Knapp and Matthews 2000). There have been a number of field observations on microhabitat use by non-indigenous fish species, such as poeciliids and cichlids, in response to predation in their indigenous and introduced ranges. Bruton and Boltt (1975) suggested that selection of shallow, well-vegetated marginal sites in Lake Sibaya, Africa, by incubating female Mozambique mouthbrooder females was to reduce risk of predation following release of the fry. They also observed nocturnal migrations of juvenile fish from shallow margins, where vegetation was sparse, to deeper water in response to presence of nocturnal predatory catfish. In the Middle Zambesi (Lake Kariba) and Upper Zambezi submerged vegetation was an important refuge for tilapiine cichlids in the presence of large mobile predators such as the characoid, *Hydrocynus vittatus* (Jackson 1961; Munro 1967; Lowe-McConnell 1982). Where large piscivores were present, tilapia populations consisted of larger fish (greater than about 17 cm TL) that were able to withstand predation, while smaller juveniles remained in shallow water among dense vegetation. Where large piscivores were absent, tilapias were observed in mixed size classes of adults and juveniles at varying depths.

The Mosquitofish and Guppy (Family Poeciliidae) in their indigenous ranges have been found to occupy shallow margins in the presence of predators (Barney and Anson 1921; Goodyear and Ferguson 1969; Goodyear 1973; Liley and Seghers 1975; Noltie and Johansen 1986), with the Guppy moving offshore into deeper waters to avoid aerial predators when aquatic predators are few or absent (Seghers 1974a, 1974b). McDowall (1996) described feral poeciliids in Australia as most abundant in warm and gently flowing or still waters in marginal areas and along the edges of aquatic vegetation. Arthington (1992) observed in urban streams in the Brisbane region, southeastern Queensland, a strong habitat preference by introduced poeciliids for the edge of pools where introduced Paragrass had invaded. Chick and McIvor (1997) found that the

Mosquitofish and Sailfin molly demonstrated preferences for different types of macrophyte cover as a refuge from predators. Winkelman and Aho (1993) found that piscivorous fish caused a shift by Mosquitofish to refuge habitats that also provided increased protection for neonates from cannibalism and predation by other species. Schramm and Zale (1985), in a study of predation on small introduced Blue tilapia (*Oreochromis aurea*) by Largemouth bass (*Micropterus salmoides*), found that survivorship of Blue tilapia was related to both effective use of protective cover and the species' morphology (deep body and spines).

Given the successful establishment of non-indigenous fishes in the Ross River, and given the presence of abundant predators, the present study investigated prey-, predator- and habitat-mediated refuges to assess their role in the establishment and persistence of non-indigenous fishes in northern Queensland waters. Very few studies have examined predator-prey interactions involving non-indigenous prey species (e.g., Mauck and Coble 1971; Schramm and Zale 1985; Chervinski *et al.* 1989; Doerner and Wagner 2003) and most have involved ambush predators, such as Northern pike (*Esox lucius*) and Largemouth bass (*Micropterus salmoides*) and one or two prey species (eg. Moody *et al*. 1983; Christensen 1996; Hayes and Wissing 1996). This is the first experimental study to examine differences in survivorship of non-indigenous fishes in mixed populations with indigenous fishes in the presence of predators with different foraging strategies (ambush or patrol) under different conditions, including prey density and levels of available cover. Within the limitations of the laboratory, these experimental conditions provided a means to obtain a detailed and more realistic assessment of survivorship of non-indigenous fishes in the presence of a predator. The non-indigenous prey species used in the study were the Mozambique mouthbrooder and the Mosquitofish. Both species are well established and widely distributed in northern Queensland (see Chapter 3) and abundant in local waters of the Townsville region, including the Ross River. The two indigenous fish species used were the Fly-specked hardyhead and Agassiz's glassperch, which are also locally abundant.

Four indigenous piscivorous fish species were used in the experiments: the Mouth almighty, Barramundi, Spangled perch and Tarpon. The experiments were designed to test the hypotheses that there were no significant differences in survivorship between the selected non-indigenous and indigenous prey species in relation to:

i. taxonomic identity;
ii. prey-mediated refuges (prey evasiveness and prey morphology);
iii. a predator-mediated refuge (predator foraging mode); and
iv. a habitat-mediated refuge (vegetation cover).

5.2 Methods

5.2.1 Prey and predator species

Prey species selection

The non-indigenous Mosquitofish (Figure 5.1a) and Mozambique mouthbrooder (Figure 5.1b) are relatively sedentary, slow moving fish, the former being deep-bodied with prominent dorsal, anal and pelvic fin spines, while the latter is narrow-bodied with soft fins. The Mosquitofish is a small species with females growing to about 6 cm and males to about 3.5 cm (McDowall 1996), while the Mozambique mouthbrooder males may grow to about 45 cm and females to about 35 cm in large water bodies such as the Ross River weirs (Webb 1994). This species may mature at a small size (<10 cm) in small ponds or pools, or when stressed, a phenomenon known as “stunting” (Noakes and Balon 1982). In this study, juvenile fish were used. The two indigenous species used in the experiments were the Fly-specked hardyhead (Figure 5.1c) and Agassiz’s glassperch (Figure 5.1d). The Fly-specked hardyhead is narrow-bodied, soft-finned and a very fast-swimming species. Agassiz’s glassperch is a deep-bodied, very mobile and manoeuvrable species with stout dorsal and pelvic fin spines. Both species are small, with the Fly-specked hardyhead growing to about 10 cm and Agassiz’s glassperch to about 8 cm (McDowall 1996; Allen, 1990,1996; Ivantsoff and Crowley 1996).

All four species have colour or morphological features that aid in predator avoidance. While countershading is a common form of concealment in fishes in open water (Chapman *et al.* 1994; Davenport and Bradshaw 1995; Stauffer *et al.* 1999), the two indigenous (fast) species (Fly-specked hardyhead and Agassiz’s glassperch) had different cryptic features (disruptive striping and transparency) associated with pelagic or schooling forms (Bond 1996; Moyle and Cech 1996; Johnsen 2000). The Fly-specked hardyhead has a horizontal mid-lateral stripe and parallel dotted lines along the body. These stripes may have a signalling function in school formation (Dafni and Diamant 1984) or help to camouflage mobile fish from predators (Hailman 1982; Rotheray 1986), while Agassiz’s glassperch is semi-transparent and laterally compressed which helps to make the fish almost invisible, particularly when only the narrowest profile is visible. The Mosquitofish, other poeciliids and several cichlid species, including the Mozambique mouthbrooder, exhibit dichromatism, a change between a pale and a dark form. This colour change has been viewed as an anti-predator response (Maglio and Rosen 1969; Casterlin and Reynolds 1977; Keita and Kohda 1995), while, in some species, it can also be used as a means of concealment from potential prey (Yamagisawa *et al.* 1990; Kohda and Michio 1993). Mosquitofish are normally pale olive-green dorsally to grey or silvery on the belly. The species

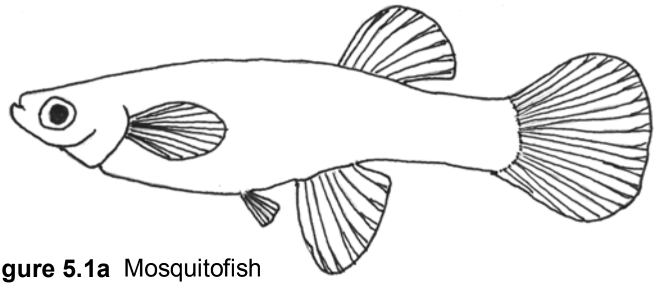


Figure 5.1a Mosquitofish

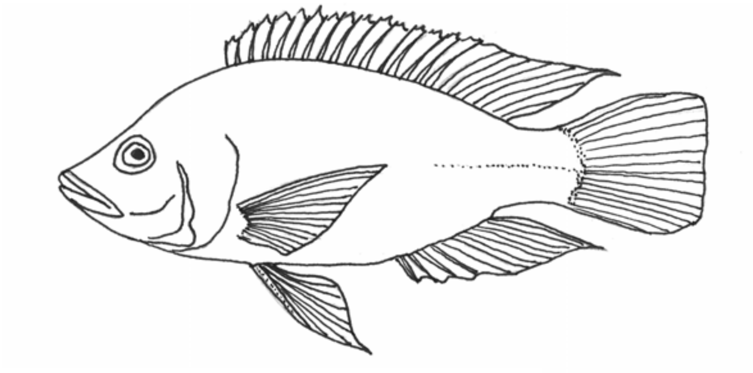


Figure 5.1b Mozambique mouthbrooder

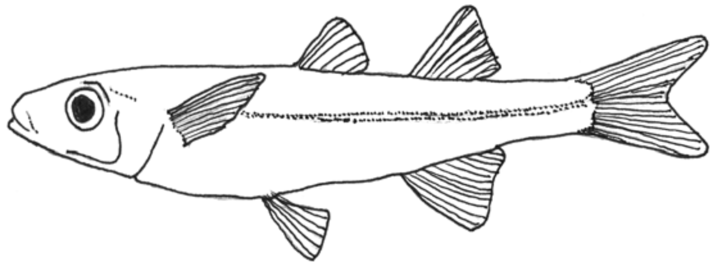


Figure 5.1c Fly-specked hardyhead

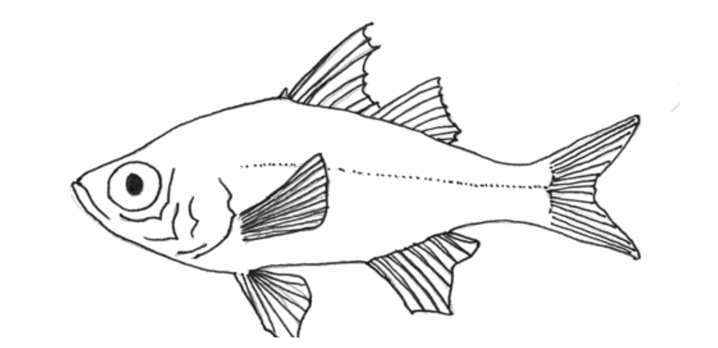


Figure 5.1d Agassiz’s glassperch

Figure 5.1 Experimental prey species can also change from a pale sandy colour when in shallow water over sandy substrate to dark olive over muddy substrate (Maglio and Rosen 1969). All four species are common in the Ross River catchment. Fish were collected from the Ross River weirs and Ross Dam using a small seine net and baited traps. The fish were then transported to the experimental facility and held in large, aerated 1,000 L oblong mesocosms for several days to acclimatise prior to the predation trials. Fish were fed daily on commercial dried flake food.

Prey morphometric data

The following morphometric data for each species were collected: total length, caudal fin width and length along the mid-line, maximum body height and maximum body width. Caudal fin length:width ratios, body height:width and length:height ratios were calculated for each species (see Appendix D, Table D1 and Figures D1 to D7). Besides body form, the four species were also classified on the basis of the type and prominence of fin support structures (spines or soft fin rays). Agassiz's glassperch and Mozambique mouthbrooder have several large, stout spines associated with the dorsal, anal and pelvic fins. The length and width of the largest pelvic and dorsal fin spines of the two species were recorded using a stereo dissector microscope with a graded objective lens (see Appendix D, Table D2 and Figures D8 to D16). A simple measure of fin-spine strength was obtained by placing a spine between thumb and forefinger and pressing together. If the spine broke it was classed as "weak"; if not it was classed as "strong". Samples of spines were tested for each size class for each species for each 1 mm body-depth size-class between 7 and 15 mm. The breakage threshold was at a spine width of 0.020 mm (see Appendix D, Figures D10 and D11, and Figures D15 and D16).

Overall, the largest dorsal and pelvic fin spines of Mozambique mouthbrooder were approximately 40% and 69% respectively of the lengths of those corresponding spines of the Agassiz's glassperch. For all body-depth size classes tested, Agassiz's glassperch dorsal and pelvic fin spines were all classed as strong (>0.020 mm width) while for Mozambique mouthbrooder, only for size classes above 15 mm body depth were all fin spines classed as strong. All Mozambique mouthbrooder less than 10 mm body depth had weak dorsal and pelvic fin spines. Between 10 and 15 mm body depth, there was a marked increase in the proportion of strong spines from 20 to 100% (see Appendix D, Figure D12). Both Mozambique mouthbrooder and Agassiz's glassperch were therefore classed as deep-bodied and spiny-finned species. Although the Fly-specked hardyhead has a number of spines in its first dorsal fin, these are very small and weak, and soft fin rays support all the other fins. This species was classified as "narrow-bodied and soft-finned" along with the Mosquitofish which has soft rays and no spines present in all fins.

Prey speed

Samples of fish of different size length classes of each of the four prey species were tested to obtain a combined relative measure of their swimming speed and manoeuvrability. Individual fish were released into a 1000 L oblong mesocosm and the time taken to capture them with a small aquarium net (approx. 10 cm x 10 cm) was recorded with a stopwatch. Fish were caught in the net without trapping against the walls of the mesocosm. A comparative index of combined swimming speed and manoeuvrability (evasiveness) (ψ) was obtained by dividing time taken to capture by total body length. Mean values of ψ for each species are included in Appendix D, Table D1 (see also Appendix D, Figures D17 to D22).

Prey classification

From morphometric and ψ data, the Fly-specked hardyhead and Agassiz's glassperch were classified as "fast" species: they have high caudal fin aspect ratios (c) which are greater than their body height-width ratios (b), and they had similar relatively high mean capture times compared with the Mosquitofish and Mozambique mouthbrooder. The ψ values indicate that, for prey of similar length, the Fly-specked hardyhead and Agassiz's glassperch took, on average, 3.5 and 4.3 times longer respectively to catch than Mozambique mouthbrooder and Mosquitofish (see Appendix D, Table D1 and Figure D22). The Fly-specked hardyhead has a moderately forked caudal fin and falcate pectoral fins which, combined with its streamlined elongate, tubular body (low b value) is characteristic of more active, faster swimming species. Agassiz's glassperch also has a forked, high aspect ratio caudal fin but has a gibbose body form (deep-bodied and laterally compressed) (high b value) which tends to reduce burst speed but increases stability and manoeuvrability (Keast and Webb 1966; Webb 1978). These differences are reflected by the relatively high ψ value (0.35) for Agassiz's glassperch compared with the "slow" species, but which is lower than the ψ value for Fly-specked hardyhead (0.56), a less manoeuvrable, but much faster species.

Both Mozambique mouthbrooder and Mosquitofish were classified as "slow" species. Their caudal fin aspect ratios were low and less than the body height-width ratios, and they had similar, low mean ψ values. The Mozambique mouthbrooder has a broad, rectangular caudal fin, large, sub-falcate pectoral fins, enlarged posterior lobes of the anal and dorsal fins and a sub-gibbose body form that is deep and moderately compressed laterally. This combination of characters is typical of manoeuvrable though slow species that often hang, or hover, in the water for long periods. (Keast and Webb 1966; Webb 1984). The Mosquitofish has a rounded, low aspect ratio caudal fin and a narrow, rounded fusiform body (low b). These characters are typical of fish, including many other cyprinodontids, that are capable of darting, evasive movements and moderate manoeuvrability, but cannot maintain very active swimming over long periods and frequently rest in mid-water or near the surface (Keast and Webb 1966).

Predator species selection

All four predator species (Barramundi, Tarpon, Spangled perch and Mouth almighty) (Figure 2.1b) occur in the Ross River catchment. Two of these species, the Barramundi and Tarpon, are catadromous and migrate into fresh water as juveniles to mature and then return to brackish estuarine and marine coastal waters to spawn. They are large species: the Barramundi may grow to about 1.8 m in length, while the Tarpon grows to about 1.5 m (Herbert and Peeters 1995; Pollard 1996). The natural occurrence of both species in the upper reaches of the Ross River has been restricted due to the presence of the barrages, although both species commonly occur in the estuarine reaches of the river, in freshwater reaches of other local, unrestricted waterways and in seasonally isolated wetlands. During the past decade, more than 70,000 Barramundi fingerlings have been stocked in the weirs and dam by the Queensland Fisheries Service (QFS) and a local fish stocking group under the Queensland State Government's recreational fishing enhancement program (see Chapter 4). Tarpon have been reported from Aplin weir, but in low numbers (Webb 1994). These fish are very strong swimmers and were probably able to move upstream and around the first barrage during floods. Their presence is also due to local anglers releasing fish caught below the first barrage into Aplin weir (personal observation). The Spangled perch and Mouth almighty are present in the Ross River catchment, although numbers have declined since the stocking of Barramundi (see Chapter 4). Unlike the Barramundi and Tarpon, the life cycles of Spangled perch and Mouth almighty are entirely in fresh water. They are medium-sized species: the Spangled perch grows to about 25 cm, while the Mouth almighty grows to about 18 cm (Pollard 1996).

Based on morphological, ecological and physiological studies of terrestrial and aquatic vertebrates including fish, these four species fit into one of two broad foraging patterns: either active patrollers or sit-and-wait ambushers (see for example, Keast and Webb 1966; Pianka 1966; Schoener 1971; Eckhardt 1979; Huey and Pianka 1981; Taigen *et al.* 1982; Webb 1978, 1984). The following morphometric data were collected for each species: total body length, caudal fin height and width, and body height and width, and gape width. A caudal fin aspect ratio (c) was calculated by fin width divided by fin length along the mid-line. A body form index (b) was calculated by maximum body height divided by maximum body width. A summary of morphometric data is presented in Appendix D, Table D1.

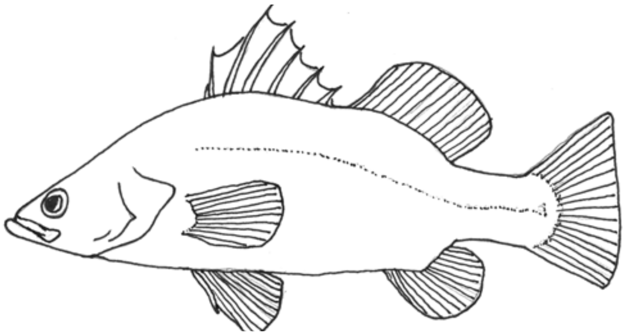


Figure 5.2a Barramundi

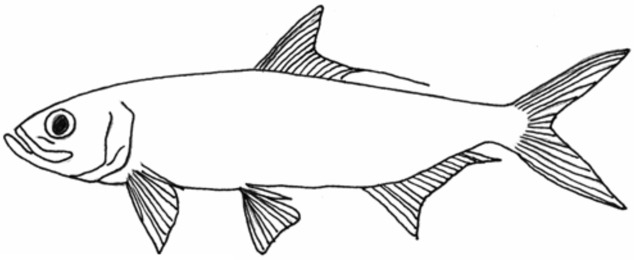


Figure 5.2b Tarpon

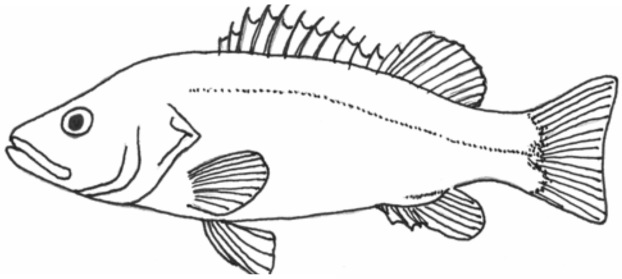


Figure 5.2c Spangled perch

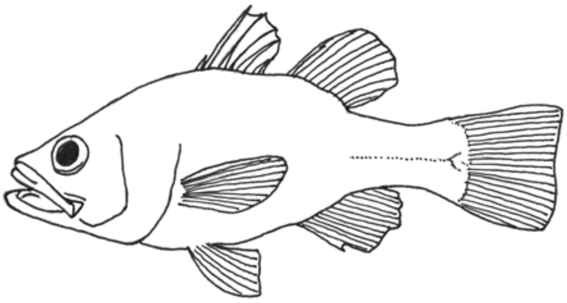


Figure 5.2d Mouth almighty

Figure 5.2 Experimental predator species

The Barramundi and Mouth almighty are sedentary, ambush predators. Typically, they have relatively low aspect ratio caudal fins and a gibbose body form. For these species, the caudal fin aspect ratio is smaller than the body height : width ratio. The caudal fin of the Mouth almighty is broad but slightly forked, while that of the Barramundi is also broad but distinctly rounded. These species are, however, capable of bursts of speed over short distances. Characteristically, both spend long periods stationary in the water column, hidden among vegetation or snags waiting for passing prey.

The Tarpon and Spangled perch are classified as "fast" cruising species with high-aspect-ratio caudal fins. The caudal fin ratio values for these species are greater than their body height:width ratios. The Tarpon has a strongly forked tail (mean aspect ratio: 6.6), with a compressed fusiform (oblong) body shape that is characteristic of very mobile species that swim more or less continuously (Keast and Webb 1966). The Spangled perch has a rounded fusiform to tubular body that is moderately deep along its length, although the body height : weight ratio is low (1.67). Its caudal fin is truncate to emarginate (weakly forked), with a moderately high mean aspect ratio of 2.8. The species is not a specialised, continuous swimmer, but is, like other percoids, an active "patroller"; it is highly manoeuvrable and also capable of bursts of speed of short to medium duration.

Mouth almighty and Spangled perch used in the study were collected from the Ross River weirs and Dam. Tarpon were collected from local watercourses on the Rowes Bay Golf course, Pallarenda, that receive floodwaters from the Bohle River, Three Mile Creek and the South Townsville Stormwater Drainage system during wet seasons. Barramundi were provided by Upper Ross Barramundi Farm, Thuringowa, as insufficient numbers of fish of the necessary size class could be caught in local waters. All fish were kept in large holding troughs several days prior to the experiments to allow time for acclimation and to ensure that fish were in good condition and feeding naturally.

5.2.2 Predator and prey size selection

The four predator species are gape-limited predators as they swallow prey whole rather than biting off pieces of prey. The maximum size of the prey consumed is therefore theoretically limited by the size of the gape or, more appropriately, the internal width of the jaws with mouth open, as predatory fish usually manipulate prey, in particular, deep-bodied prey with fin spines, so that the fish are swallowed head first and on their side (e.g., Hoogland *et al.* 1956; Hoyle and Keast 1988). Appropriate prey (body depth and length) and predator size (gape width) for the mixed prey population survivorship trials were selected using prey survivorship data from experiment 1 with Mouth almighty as predator and Mozambique mouthbrooder as prey. A prey electivity index (ε_i) was calculated as devised by Hambright (1991), which is a modified form of the Manly-Chesson preference index α_I (Manly 1974; Chesson 1983).

Values of α for each prey size class were calculated for each feeding trial as:

$$\alpha_i = \ln[(n_{i0} - r_i)/ n_{i0} \;/ \sum_{j=1}^{m} \ln[(n_{j0} - r_j)/ n_{j0}]$$

where $i = 1, 2, \ldots\ldots, m$; m is the number of prey size-classes, r_i is the number of individuals of prey size-class i ingested, and n_0 is the number of individuals of prey size-class i present at the beginning of a feeding trial, and

$$\varepsilon_i = (m\alpha_i - 1)/[(m - 2)\alpha_i + 1].$$

The predator electivity index ε_i ranges from +1 representing total preference (all prey of the size class i eaten) to –1 representing total avoidance (all prey of the size class i not eaten), with 0 representing no preference. Index values for each predator size class were plotted (see Results section 5.3, Figures 5.3a to 5.3i).

Values of the prey body-depth to predator gape-width ratio (BD:GW) for each predator size class using BD_{max} values when $\varepsilon > +0.5$ (i.e., where a strong positive prey preference was indicated) were plotted (see Results section 5.3, Figure 5.5) and a Pearson correlation coefficient calculated. The mean BD_{max} was 10.33 mm (10 mm) and was used as the experimental prey body depth (for deep-bodied fish). A regression line with the equation: $BD = 6.591 + 0.236GW$ was fitted to the plot of BD_{max} at $\varepsilon_{(> +0.5)}$ against predator gape-width class. The predator gape width that corresponded to the mean BD_{max} was 15.85 mm (16 mm) and was then used as the experimental predator gape-width value. Mean prey body depth against predator gape width was also plotted and a correlation coefficient calculated. Plots of prey body depth and length against predator gape width and length from samples of Mouth almighty and Barramundi collected from the Ross River weirs were also done for comparison with the experimental data obtained for Mouth almighty.

For the survivorship trials, since Agassiz's glassperch and Mozambique mouthbrooder have very similar l : h ratios, a body-depth of 10 mm was used for both species. The narrow-bodied fish used in the trials, the Mosquitofish and Fly-specked hardyhead, were then selected to be within the same total length range (30 to 40 mm) as Agassiz's glassperch and Mozambique mouthbrooder with a body depth of 10 mm.

5.2.3 Survivorship trials

The mesocosms for all trials were aerated 1000 L oblong troughs fitted with shadecloth lids. In experiment 1, the prey species used was the non-indigenous Mozambique mouthbrooder and the indigenous predator species used was the Mouth almighty. In experiment 2, 3 and 4, mixed prey populations of two indigenous species (Fly-specked hardyhead and Agassiz's glass perch) and two non-indigenous species (Mozambique mouthbrooder and Mosquitofish) were established in the mesocosms prior to introduction of selected predators from holding tanks. The predators used were the Barramundi and Mouth almighty (ambush predators) and Tarpon and Spangled perch (patrol predators). Prey fish were fed daily *ad libitum* with dried flake food. Observations of predator and prey behaviour were made at intervals during each experiment.

Experiment 1: Survivorship of juvenile Mozambique mouthbrooder in relation to prey body depth and gape width of the predatory Mouth almighty

Populations of the Mozambique mouthbrooder, consisting of five individuals for each 1 mm body-depth size class between 7 mm and 23 mm, were placed in each of three mesocosms with no vegetation cover. Three Mouth almighty of the same gape width (GW) were then placed in each mesocosm. At the end of three days, the number of Mozambique mouthbrooder remaining in each size class was recorded. The experiment was repeated using Mouth almighty of different gape-width classes. Nine predator size classes were selected which ranged from TL: 7 cm (GW: 10 mm) to TL 15 mm (GW: 23 mm). The gape width was measured using callipers and taken as the internal distance across the lower jaw of the fish with its mouth open. Morphometric data for Mouth almighty (and Barramundi) and for prey items present in gut samples from fish collected in the Ross River weirs (see Chapter 4) were used for comparison with data collected in this experiment.

Experiment 2: Survivorship of non-indigenous fish species in mixed populations with indigenous species with no cover and in the presence of ambush or patrol predators

Twenty individuals of each of four selected prey species were placed in each of three mesocosms. Three individuals of one predator species from holding tanks were placed in each mesocosm. There were three "time" treatments (24hrs, 48hrs, and 72 hrs) with three replicates per treatment. At the end of three days, the number of survivors of each prey species was recorded.

Experiment 3: Survivorship of non-indigenous fish species in mixed populations with indigenous species with one dominant prey taxon and no cover, in the presence of ambush or patrol predators

In this experiment, prey populations consisted of one "dominant" species (150 individuals) with the other three species as "non-dominants" (10 individuals each species). There were three replicates for each treatment (prey density) with two different treatments; that is, the number of each prey species was, in turn, increased to 150 with the other three species kept at 10 individuals each. Three individuals of one predator species from holding tanks were introduced into each mesocosm. At the end of three days, the number of survivors of each prey species was recorded.

Experiment 4: Survivorship of non-indigenous fish species in mixed populations with indigenous species at equal densities, with different levels of cover in the presence of ambush or patrol predators

Prior to the experiment, the submerged aquatic macrophyte, Hornwort, *Ceratophyllum demersum,* was collected from the Ross River. This plant is a dominant component of the aquatic vegetation in the river weirs and Ross Dam, and occurs in very dense patches, especially in shallower margins. The vegetation collected was placed in a large holding tank and well rinsed prior to use in the experimental mesocosms. For the medium density cover, 4.0 kg of vegetation (wet weight) were used, providing approximately 40-50% coverage by volume below water of the mesocosm. The vegetation was randomly located in the trough to form patches with several areas of open water. For the high-density cover, 8.0 kg of vegetation (wet weight) were used, providing approximately 90% coverage by volume below water and left a few, small open patches of water in each mesocosm. Additional aeration was provided for all treatment levels, with aerators placed in the open patches of water in the high density cover treatment. Twenty individuals of each of the four prey species were placed in each mesocosm with three individuals of one predator species. There were three treatments: no vegetation cover, medium cover and high cover. Observations were made of the behaviour of predator and prey, and after 72 hours, the number of survivors of each prey species was recorded. The experiment was repeated using each of the four predator species.

5.2.4 Data analysis

Experiment 1 see Section 5.2.2

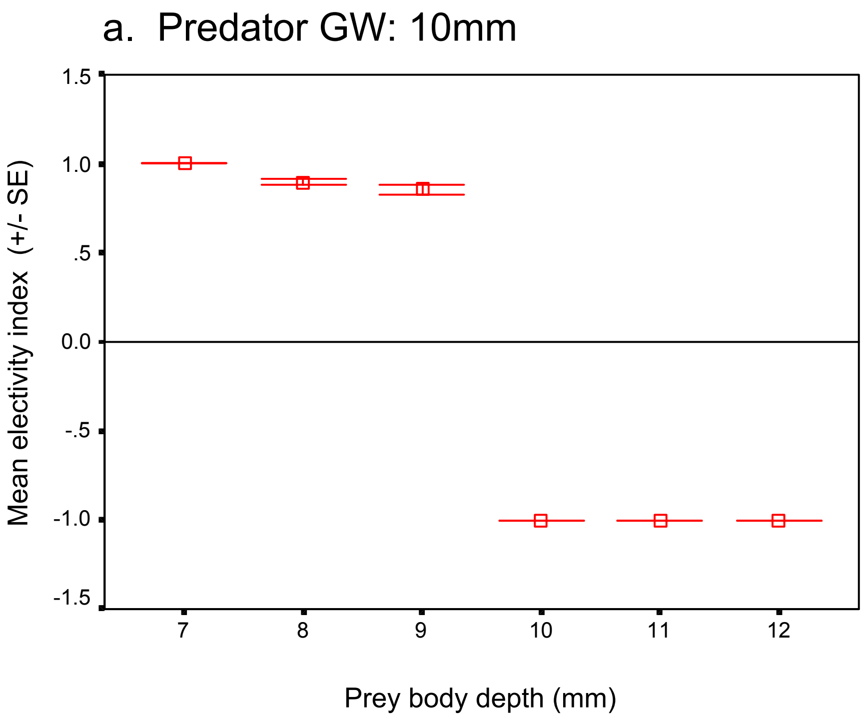
Experiment 2, 3 and 4 ANOVA analyses were performed on data sets transformed by the arcsine transformation $[\sin^{-1}(\sqrt{x})]$ for proportional (% survivorship) data (Fowler *et al.* 1990). Where appropriate, the Bonferonni adjustment to the alpha level was made for non-normal data or multiple comparisons. For comparison of prey survivorship between predator classes, combined data sets were used for ambush predators (Barramundi and Mouth almighty) and for patrol predators (Tarpon and Spangled perch). All predator data sets were combined for analysis of prey survivorship in the presence of a predator irrespective of foraging mode.

5.3 Results

5.3.1 Survivorship of juvenile Mozambique mouthbrooder in relation to gape-width of the predatory Mouth almighty.

Plots of mean ε_i of each prey size-class for each predator size-class are presented in Figure 5.3 In all cases, prey larger than the gape width of the predator ($\varepsilon = -1$) were not ingested. The maximum prey size ingested by Mouth almighty less than 15 mm gape width was between 89.6 and 93.3% of their gape width. For Mouth almighty with gape widths greater than 15 mm, the maximum prey body depths were considerably smaller – between 65.2 and 84.8% of the predator gape width. The marked decrease in the ratio values corresponds with prey body depths greater than 15 mm when all Mozambique mouthbrooder fin spines were classed as “strong” (Appendix D, Figure D12).

Using BD_{max} values there was a strong negative correlation ($r = 0.868$) between the BD:GW ratio and predator size (Figure 5.4) and a strong correlation ($r = 0.736$) between prey body depth and predator gape width (Figure 5.5). While prey size increased with increasing size of the predator, larger predators selected proportionately smaller prey relative to their gape width. The mean maximum prey body depth, where $\varepsilon > 0.5$, was 10.33 mm. From the regression equation: $BD = 6.591 + 0.236GW$, for $BD = 10.33$ mm, the predator gape width was 15.85 mm. For this combination of prey and predator sizes, the overall “optimum” value of the BD:GW ratio was



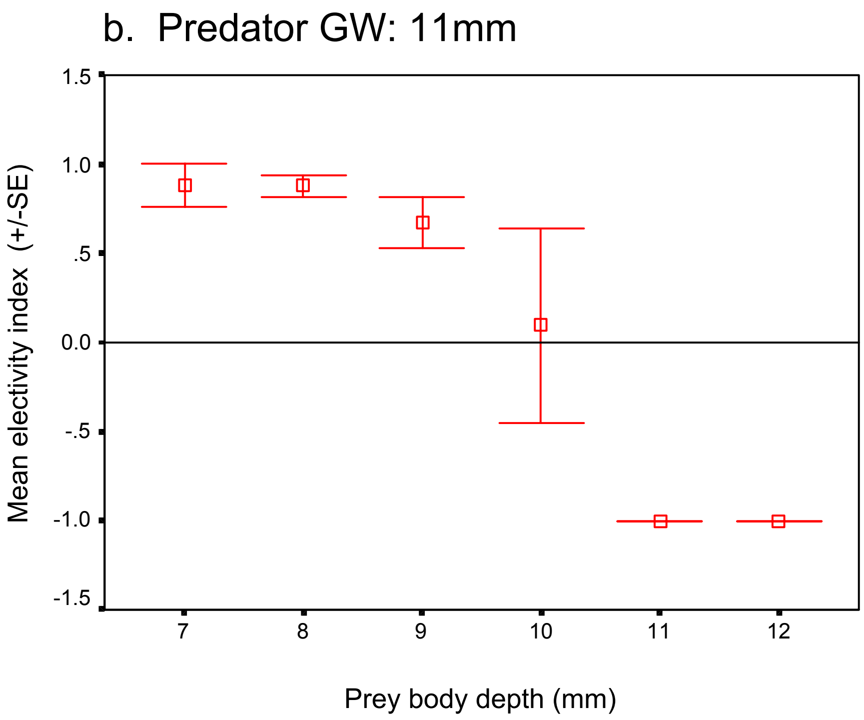
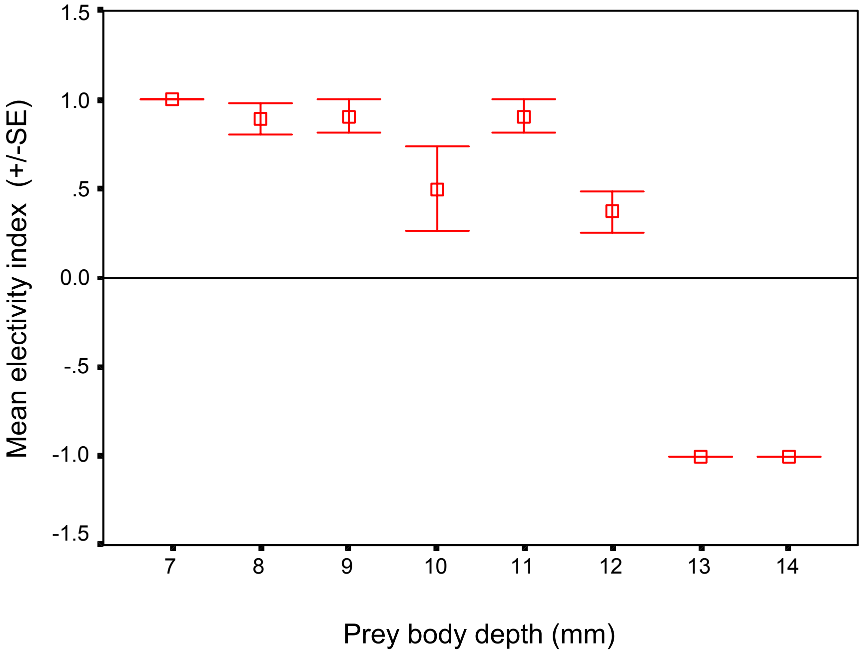
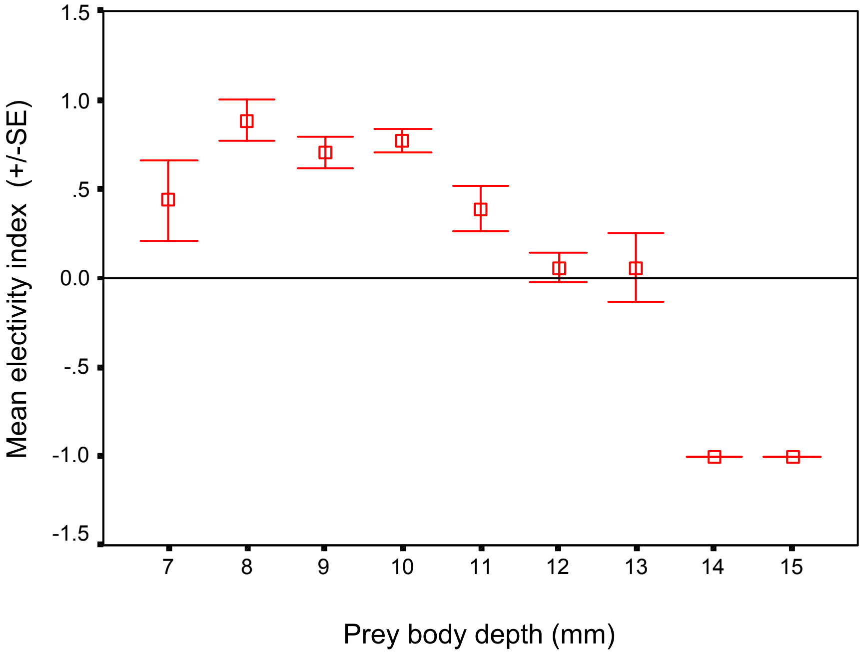


Figure 5.3 Mean electivity index for different prey (Mozambique mouthbrooder) body-depth classes for predator (Mouth almighty) gape-width class: a, 10 mm, b, 11 mm, c, 13 mm, d, 14.5 mm, e, 15 mm, f, 16.5 mm, g, 19mm, h, 21mm, i, 23mm

continued



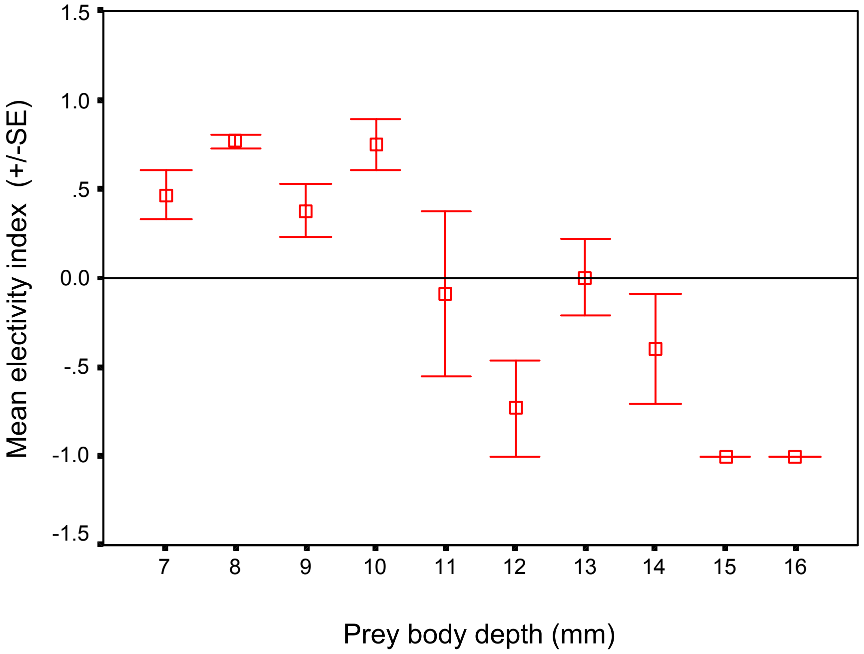
c. Predator GW: 13mm



d. Predator GW: 14.5mm

Figure 5.3 continued

e. Predator GW: 15mm



f. Predator GW: 16.5mm

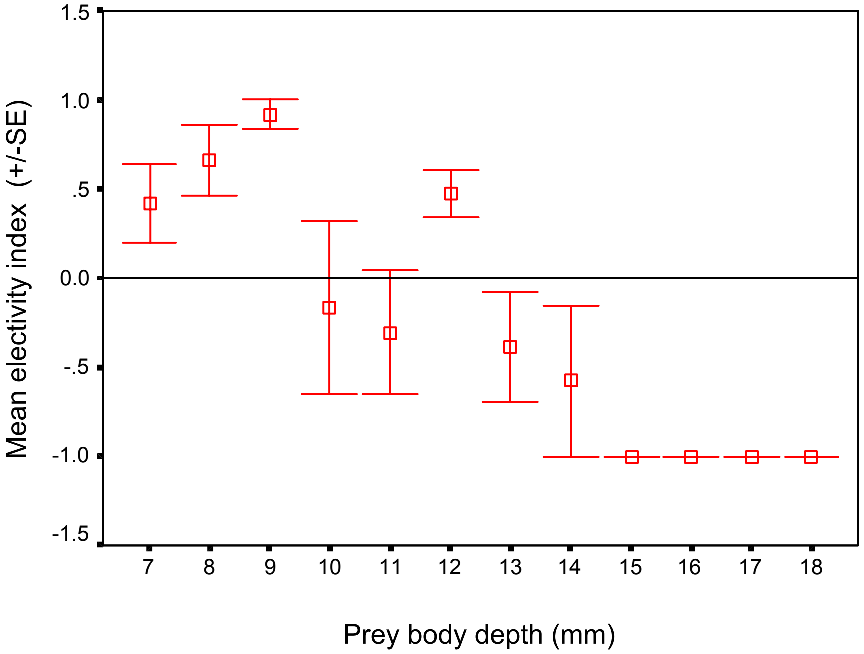
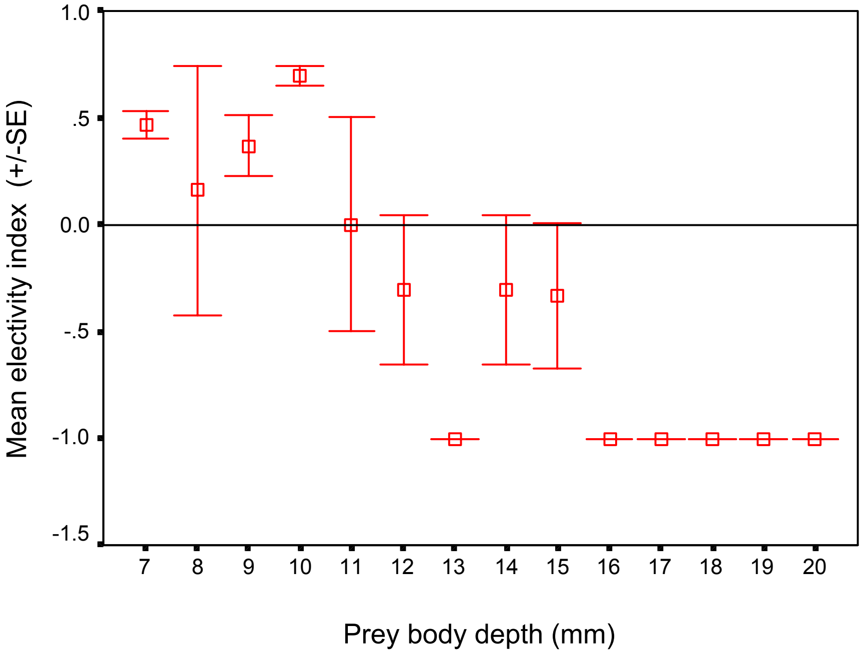
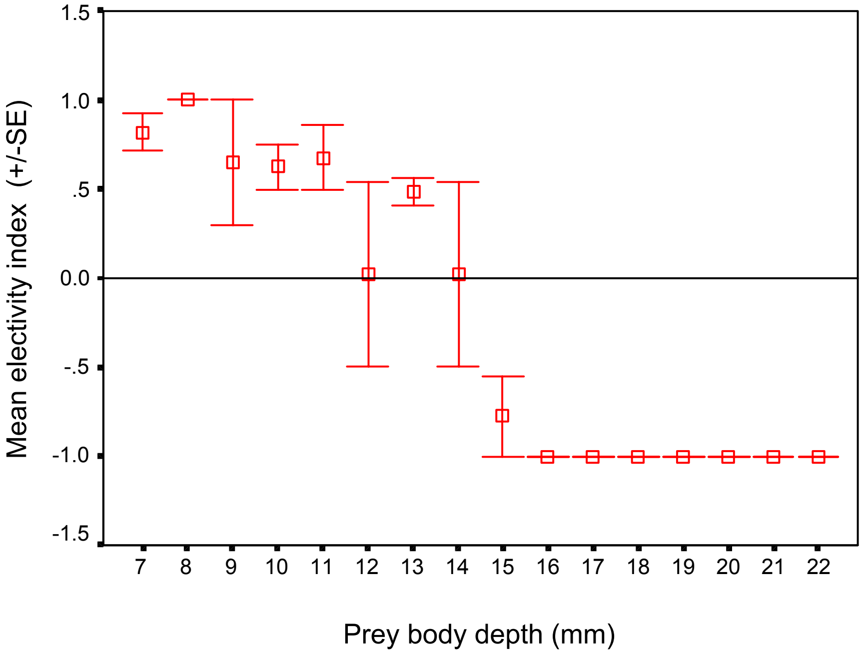


Figure 5.3 continued



g. Predator GW: 19mm



h. Predator GW: 21mm

Figure 5.3 continued

i. Predator GW: 23mm

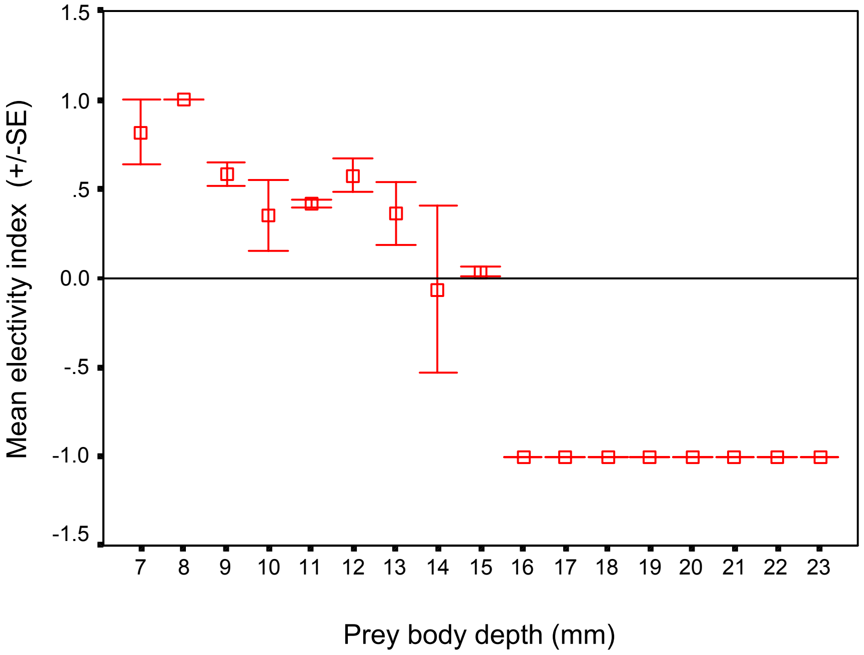


Figure 5.3 continued

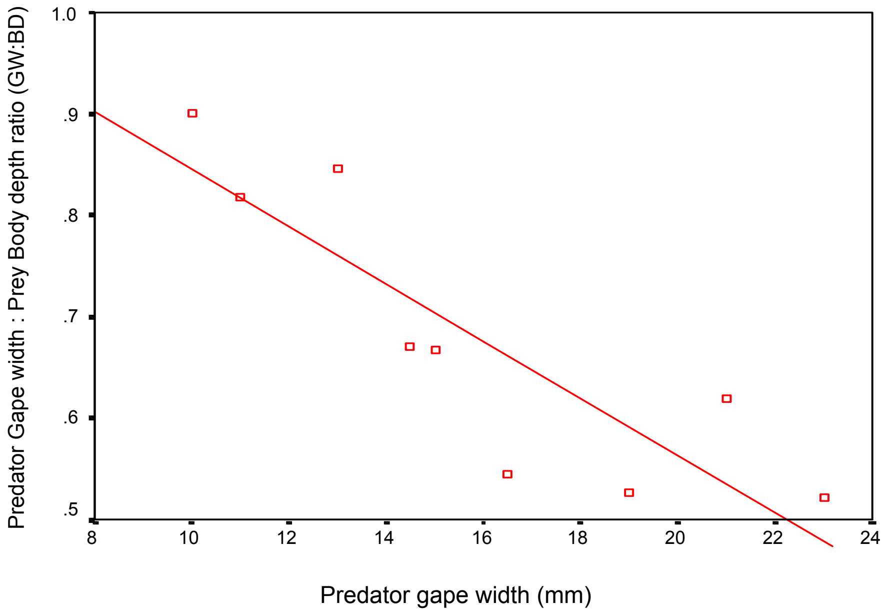


Figure 5.4 Predator (Mouth almighty) gape width : Prey (Mozambique mouthbrooder) body depth ratio *v* Predator (Mouth almighty) gape width

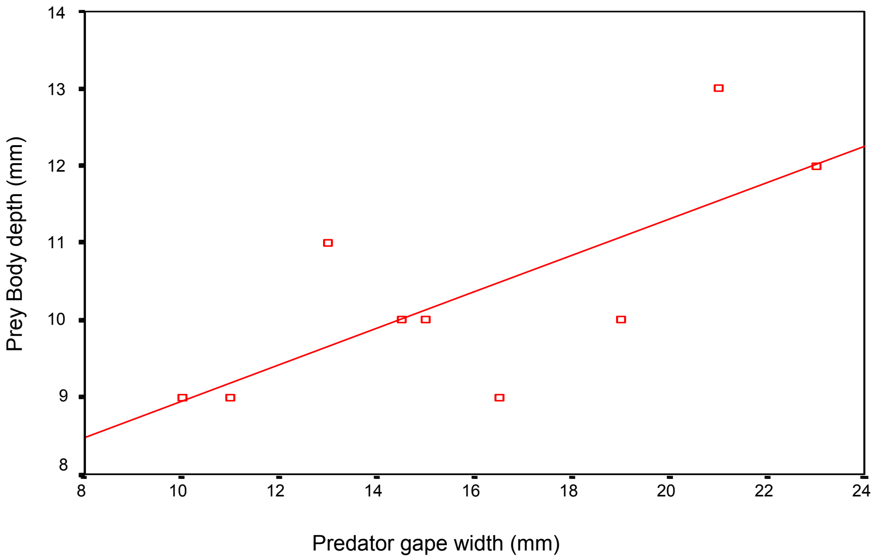


Figure 5.5 Prey (Mozambique mouthbrooder) body depth *v* predator (Mouth almighty) gape width

0.652. The mean GW:BD values calculated for each size class ranged from a maximum of 0.79 (Pred. GW= 10 mm) to a minimum of 0.44 (Pred. GW = 23 mm) (Table 5.1) with an overall mean value of 0.613 similar to the value obtained using the BD_{max} versus Gape-width regression equation.

Predator and prey morphometric field data: Mouth almighty

A total of 318 prey items (fish and invertebrates) were measured from 193 Mouth almighty collected from sites in the main channel of the Ross River Weirs, Ross Dam and Upper Ross. A summary of the morphometric data for Mouth almighty and prey items is presented in Table 5.1. Mouth almighty ranged in size from 3.1 to 14.4 cm (TL) with gape widths ranged between 0.4 and 1.8 cm. For small Mouth almighty, prey items ranged in size from 0.5 to 3.0 cm (TL) and 0.1 to 0.5 cm (BD). For medium-sized Mouth almighty, prey items ranged in size from 0.8 to 4.8 cm (TL) and 0.1 to 0.9 cm (BD). For large Mouth almighty, prey items ranged in size from 1.0 to 4.3 cm (TL) and 0.2 to 0.9 cm (BD). There was a strong correlation between gape width and total body length for the Mouth almighty (r^2=0.921) (Figure 5.6) and a modest correlation between prey body depth and predator gape width (r = 0.489) (Figure 5.7) and between prey length and predator body length (r = 0.435) (Figure 5.8). Overall, the median prey body depth was 33% of the predator gape width, with a range of 12 to 64%. There was a weak, but significant, negative correlation (-0.254) between prey body depth to predator gape-width ratio and predator length (BD:GW-TL) (Figure 5.9), and a slightly stronger negative correlation (r = -0.40) when only fish prey were included.

A summary statistical analysis of prey size in relation to predator size class (small, medium and large fish) is presented in Table 5.1. While overall prey size increased with increasing predator size, small Mouth almighty consumed relatively larger prey in relation to their gape width compared with the medium and large size classes (Figure 5.9). Among all predator size classes (small, medium, large) there were significant differences in median body length of prey consumed (1.8, 2.0, 2.5 cm respectively) and in median body depth of prey consumed (0 .3, 0.4, 0.5 cm respectively) (Figures 5.10 and 5.11). Small Mouth almighty had a significantly larger GW:BD ratio (0.36) compared with larger size classes, while there was no significant difference between GW:BD ratios for medium-sized (0.32) and large-sized (0.29) Mouth almighty (Figure 5.12). Mean GW:BD values, including invertebrate and vertebrate prey, are also presented in Table 5.2. Mean ratio values for fish prey only were for small Mouth almighty: 0.41 (range: 0.24-0.62); medium-sized Mouth almighty: 0.34 (range: 0.18-0.64) and large-sized Mouth almighty: 0.27 (range: 0.15-0.44). Overall, the mean GW:BD ratio for the total sample for fish

Table 5.1 Summary of field and experimental morphometric data analyses for the predatory Mouth almighty and its prey

LABORATORY MORPHOMETRIC DATA
Mouth almighty (predator) – Mozambique mouthbrooder (prey)

GW(mm)	10	11	13	14.5	15	16.5	19	21	23
Mean GW:BD	0.79	0.68	0.70	0.66	0.63	0.57	0.52	0.53	0.44
MAX (mm)	9	10	12	13	14	14	15	15	15
MAX/GW (%)	90.0	90.9	92.3	89.6	93.3	84.8	78.9	71.4	65.2

FIELD MORPHOMETRIC DATA
Mouth almighty (predator) and Prey (gut contents)

E.		Min	Max	mean	sd	median	IQ	r^2
F. Pred.	(TL) small	3.1	6.9	6.0	0.886	6.3	0.70	
	med.	7.0	9.9	8.1	0.850	8.0	1.21	
	large	10.3	14.4	11.3	0.776	14.4	0.70	
Pred.	(GW) small	0.4	1.0	0.8	0.143	0.8	0.30	
	med.	0.8	1.7	1.1	0.147	1.1	0.25	
	large	1.4	1.8	1.7	0.135	1.7	0.20	
Pred.	TL-GW							0.921
Prey$_{pred}$	(TL) small	0.5	3.0			1.8	0.50	
	med.	0.8	4.8			2.0	0.72	
	large	1.0	4.3			2.5	1.00	
Prey$_{pred}$	(BD) small	0.1	0.5			0.3	0.05	
	med.	0.1	0.9			0.4	0.02	
	large	0.2	0.9			0.5	0.10	
Prey:Pred	BD: GW (all prey)	0.64	0.12	0.34	0.102	0.33	0.10	0.489
Prey$_{pred}$	small	0.2	0.6	0.4	0.103	0.3	0.13	
	med.	0.1	0.6	0.3	0.096	0.3	0.12	
	large	0.1	0.5	0.3	0.094	0.2	0.15	
Prey-Pred	TL-GW (all prey)							0.388
Prey-Pred	TL-TL							0.435
All prey	BD:GW-Tl$_{pred}$							-0.254

FIELD MORPHOMETRIC DATA
Mouth almighty GW:BD ratio – predator size class comparison

	Predator size class	Chi Sq.	p. (0.05)
Prey total length (cm)	small-medium	19.724	<0.001
	medium-large	13.955	<0.001
Prey body depth (cm)	small-medium	25.192	<0.001
	medium-large	25.937	<0.001
GW:BD	small-medium	15.898	<0.001
	medium-large	1.424	0.233

GW: predator gape-width; GW:BD : predator gape width to prey body-depth ratio; MAX: maximum prey body depth size ingested by each predator size class; MAX/GW : Maximum prey body depth as a percentage of predator gape-width for each size class of predator

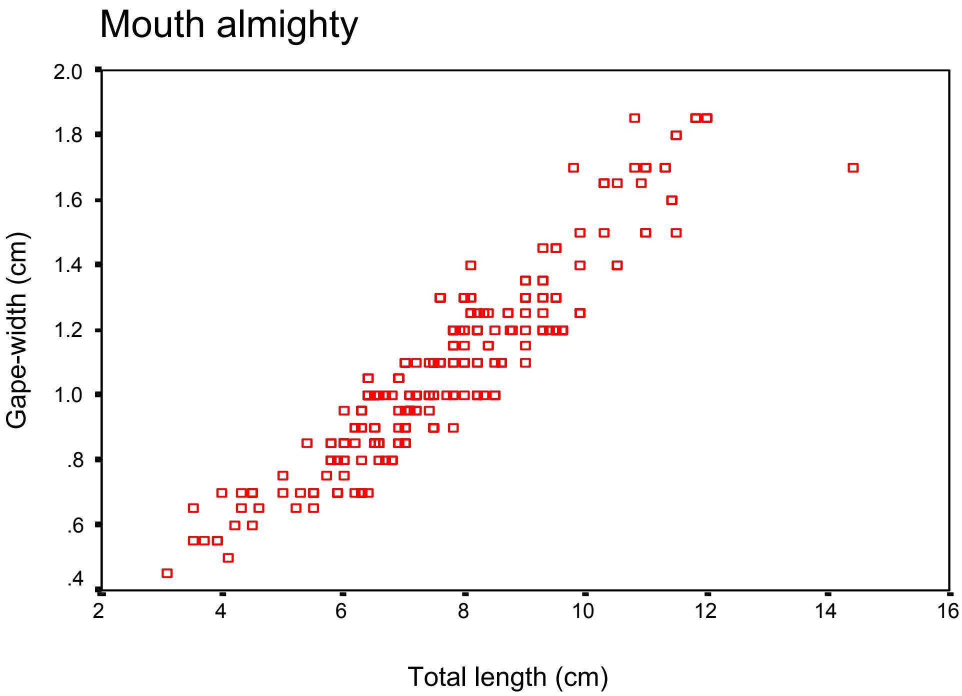


Figure 5.6 Gape width *v* total body length for the predator, Mouth almighty

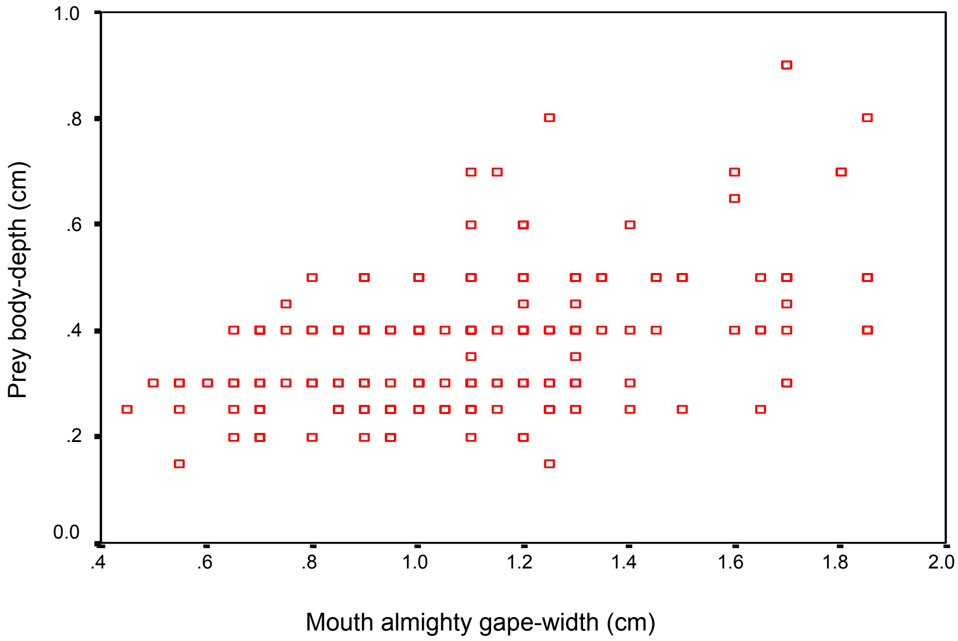


Figure 5.7 Prey body depth *v* predator gape width for samples of Mouth almighty

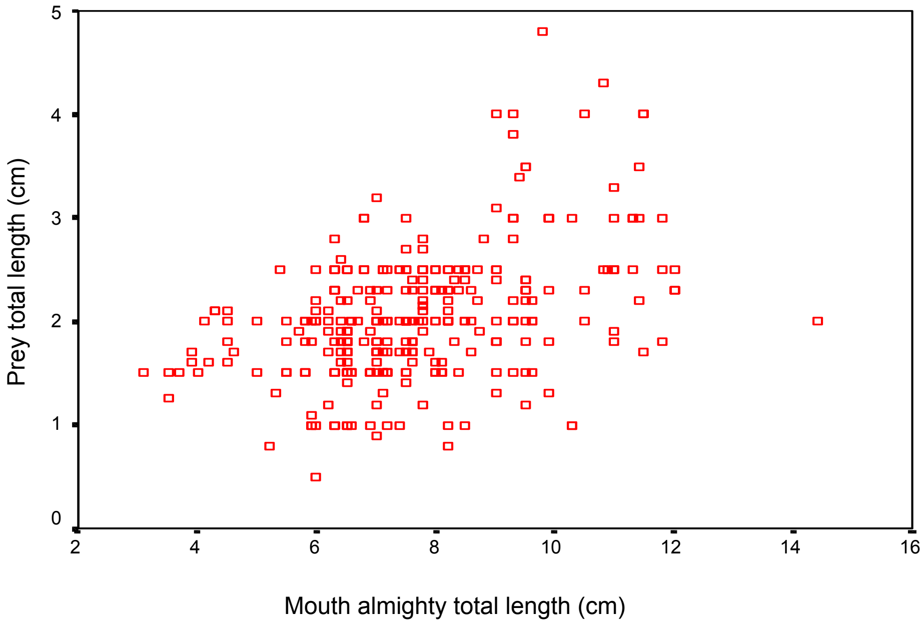


Figure 5.8 Prey total length *v* predator total length for samples of Mouth almighty

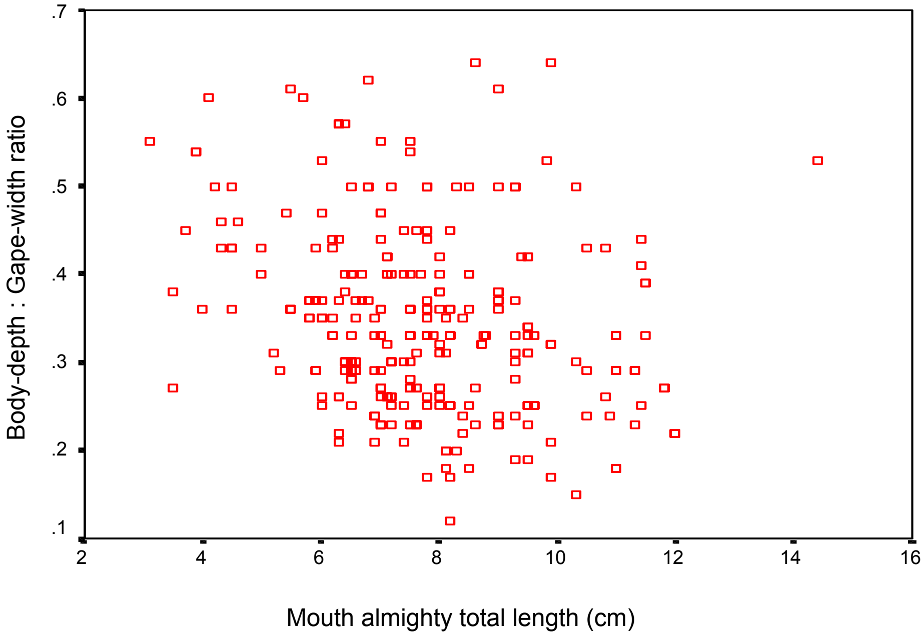


Figure 5.9 Prey body-depth to Predator gape-width ratio (BD : GW) *v* Predator body length for samples of Mouth almighty

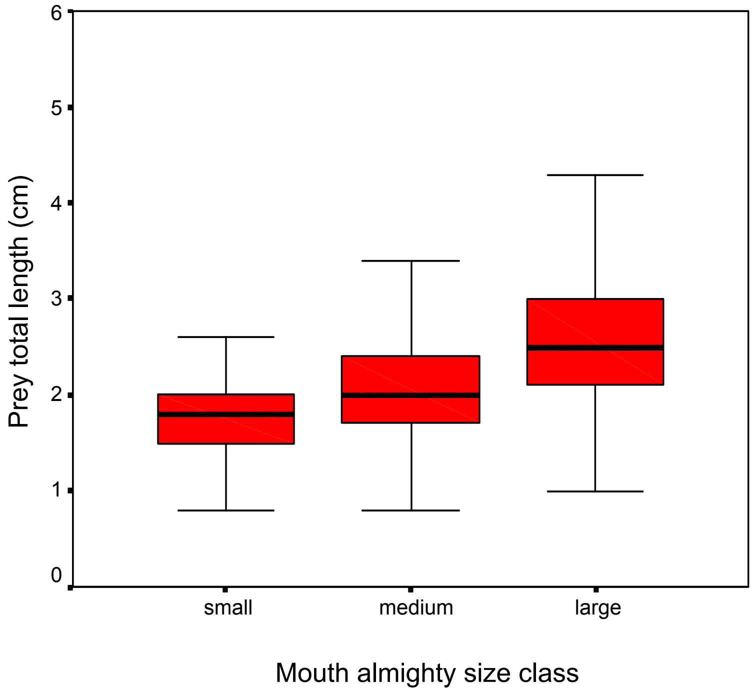


Figure 5.10 Prey median total length for each size class of Mouth almighty

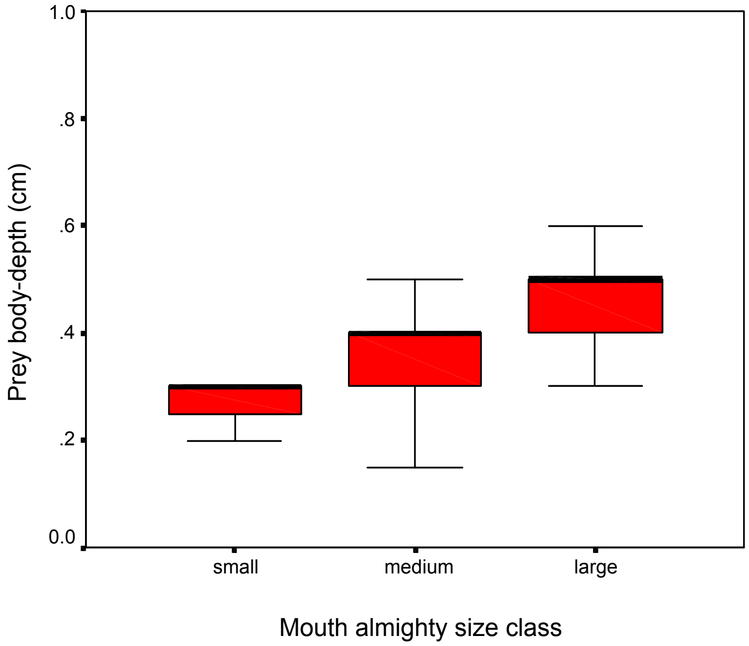


Figure 5.11 Prey median prey body depth for size classes of the predatory Mouth almighty

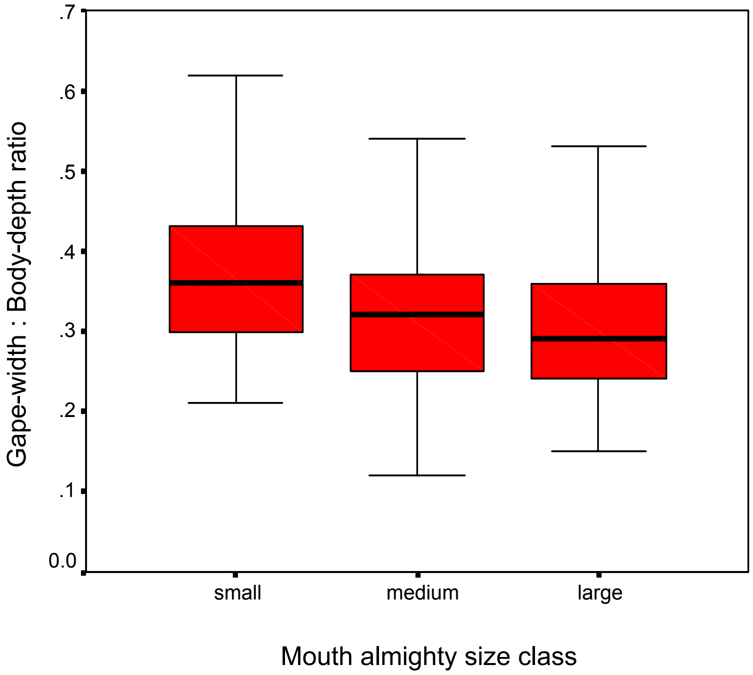


Figure 5.12 Median gape-width to prey body-depth ratio (GW : BD) for size classes of the Mouth almighty

prey only was 0.35 (all prey items: 0.34), with a range of 0.15 to 0.64 (all prey items: 0.12-0.64).

Predator and prey morphometric data: Barramundi

Barramundi collected from the weirs ranged in size from 31 to 100 cm (TL). Their gape width ranged from 2.7 to 8.8 cm. The mean prey body length was 4.62 cm (s.d. 2.242) with a range from 1.2 cm to 17.0 cm. Prey body depth ranged from 0.2 cm to 4.5 cm. The mean predator Gape-Width to prey Body-Depth ratio (GW:BD) expressed as a percentage was 20.3% with a range from 4.2 to 78.1% (Table 5.2). There was a strong correlation between total body length and gape width for Barramundi (r^2 = 0.844) (Figure 5.13). There was a weak, but significant correlation between predator body length and prey body length (Table 5.2 and Figure 5.14) and between predator gape width and prey body depth (Figure 5.15). There was a very weak, though not significant, negative correlation between predator body length and the GW:BD ratio (Figure 5.16).

Table 5.2 Morphometric summary for Barramundi and prey items of fish collected from the Ross River Weirs, 1997/98

Barramundi	**min.**	**max.**	**mean**	**s.d.**	**r**	**p. (0.01)**
$TL_{predator}$ (cm)	31	100	56.6	12.682	-	-
$GW_{predator}$ (cm)	2.7	8.8	5.5	1.137	-	-
TL_{prey} (cm)	1.2	17.0	4.6	2.242	-	-
BD_{prey} (cm)	0.2	4.5	1.1	0.714	-	-
BD:GW	0.042	0.781	0.203	0.134	-	-
TL-GW $_{predator}$	-	-	-	-	0.844	<0.001
$TL_{predator}$-TL_{prey}	-	-	-	-	0.241	<0.001
BD_{prey}-$GW_{predator}$	-	-	-	-	0.230	<0.001
$TL_{predator}$- BD:GW	-	-	-	-	-0.034	0.441

TL=total length; GW=gape width; BD= body depth

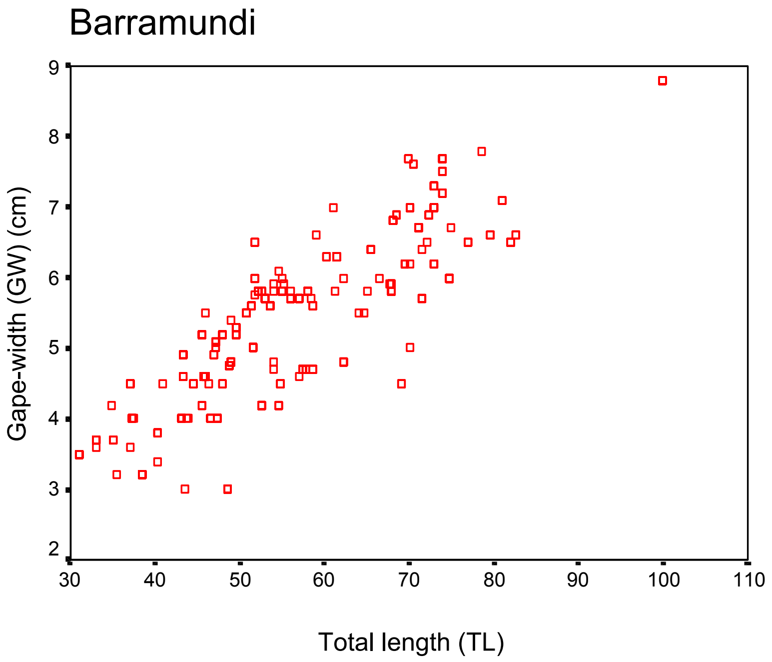


Figure 5.13 Gape width *v* total body length for the predatory Barramundi

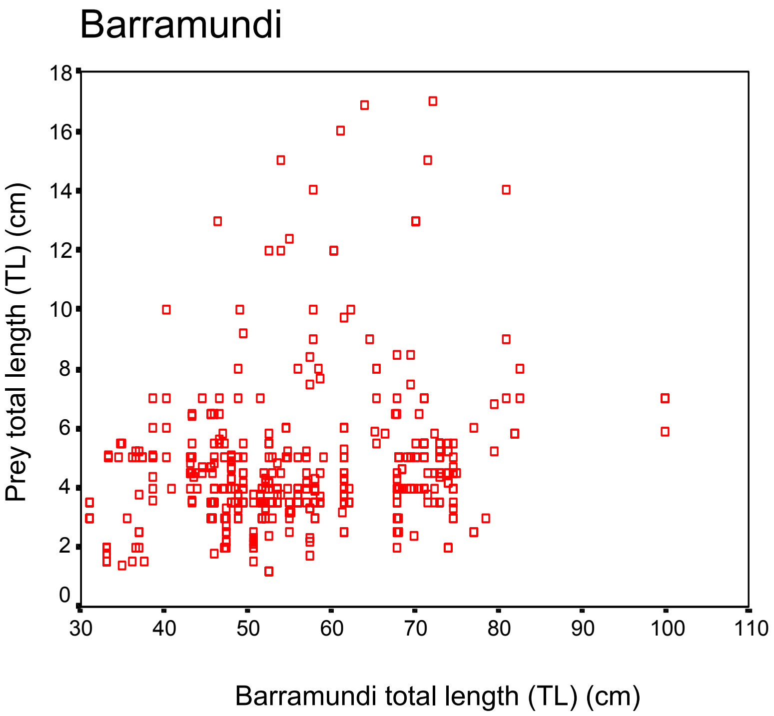


Figure 5.14 Prey body depth *v* gape width of the Barramundi

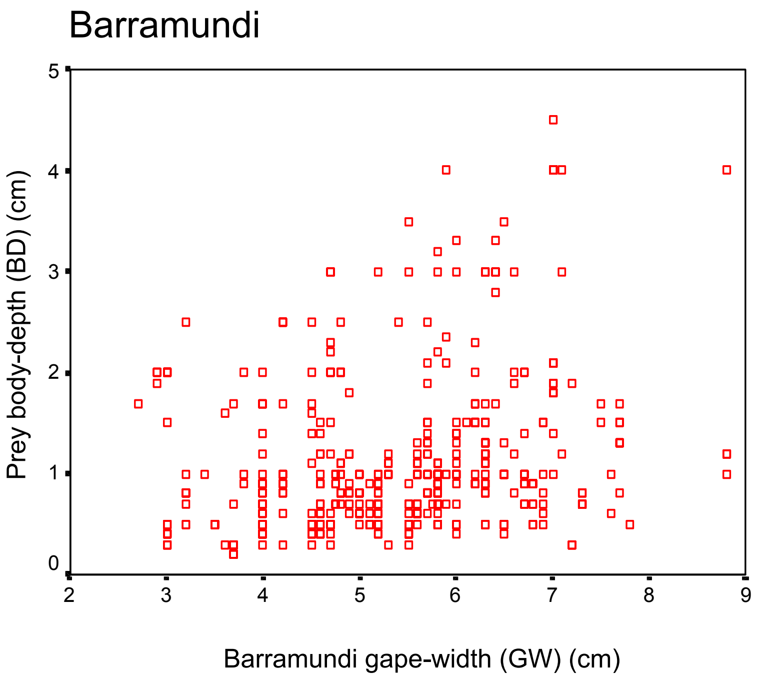


Figure 5.15 Prey body depth to predator gape width ratio (BD:GW) *v* total body length for the Barramundi

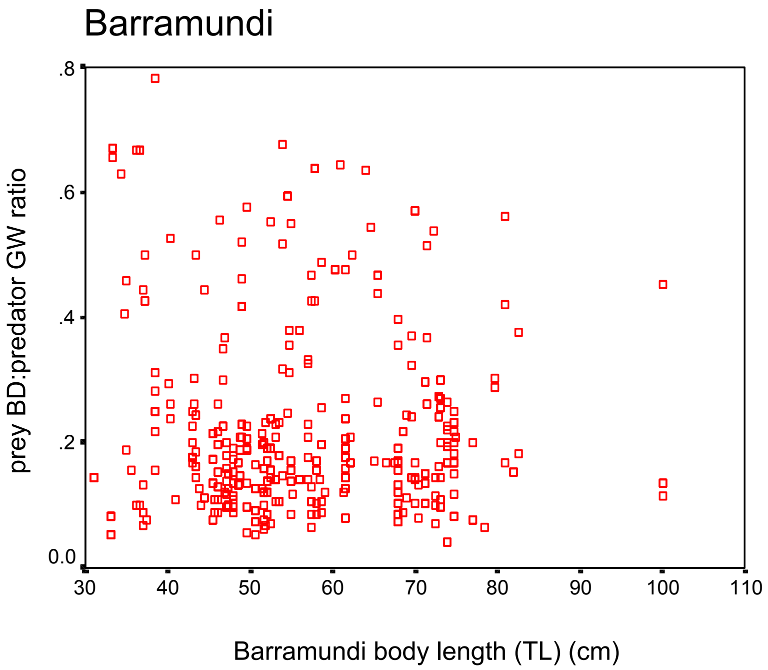


Figure 5.16 Prey body depth to predator gape width ratio *v* total body length for Barramundi

5.3.2 Experiment 2: Survivorship of non-indigenous fish species in mixed prey populations

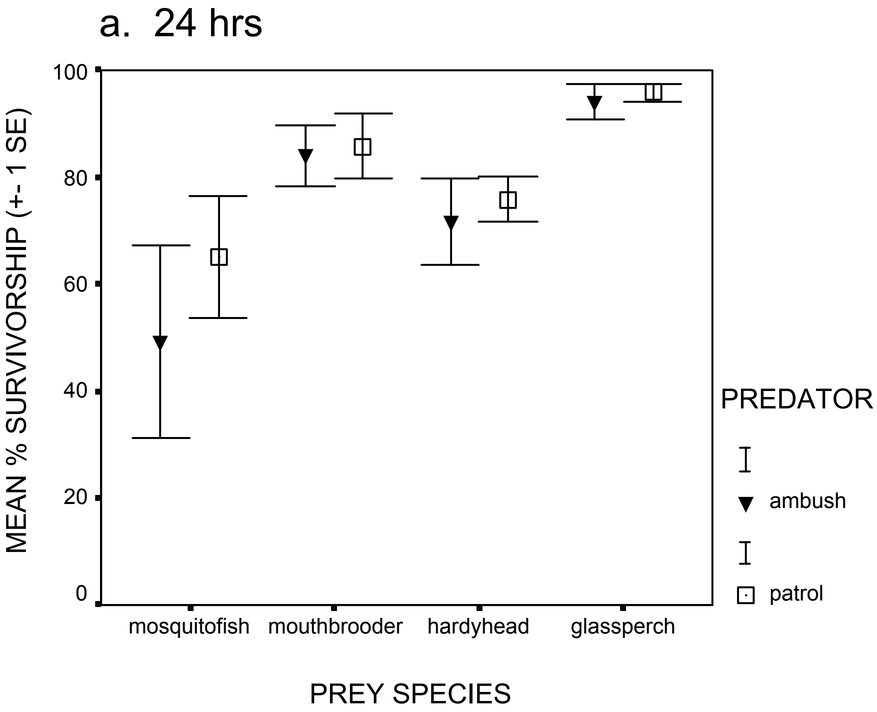
The results showed that there were no significant differences in mean survivorship for each of the four prey species between ambush predators and patrol predators at each time interval and no interaction effects between all independent variables (Table 5.3 and Figure 5.17). There were no significant differences in mean survivorship for prey classes (narrow body, deep body, slow, fast) at each time interval between ambush predators and patrol predators (Table 5.4 and Figure 5.18).

Table 5.3 Statistical summary for comparison of prey species survivorship between ambush and patrol predators at different time intervals

Hrs	Prey spp	Ambush predators		Patrol predators	
		% survivorship		% survivorship	
		mean	se	mean	se
24	Mosquitofish	49.3	15.17	68.0	13.56
	Mouthbrooder	84.2	5.69	85.8	5.97
	Hardyhead	71.7	8.23	75.8	4.17
	Glassperch	94.2	3.27	95.8	1.54
48	Mosquitofish	30.7	14.49	22.0	12.41
	Mouthbrooder	51.7	19.00	75.8	7.46
	Hardyhead	44.2	20.02	42.5	7.93
	Glassperch	68.3	13.88	74.2	12.94
72	Mosquitofish	7.9	4.48	0.0	0.00
	Mouthbrooder	8.3	4.01	27.5	12.37
	Hardyhead	43.3	19.44	19.2	7.00
	Glassperch	57.5	18.34	40.3	14.53

ANOVA

	df	F	P (0.01)
Species	3	14.233	<0.001
Predators	1	0.108	0.744
Time	2	34.375	<0.001
Species * Predator	3	0.591	0.622
Species * Time	6	0.780	0.587
Predator * Time	2	0.736	0.481



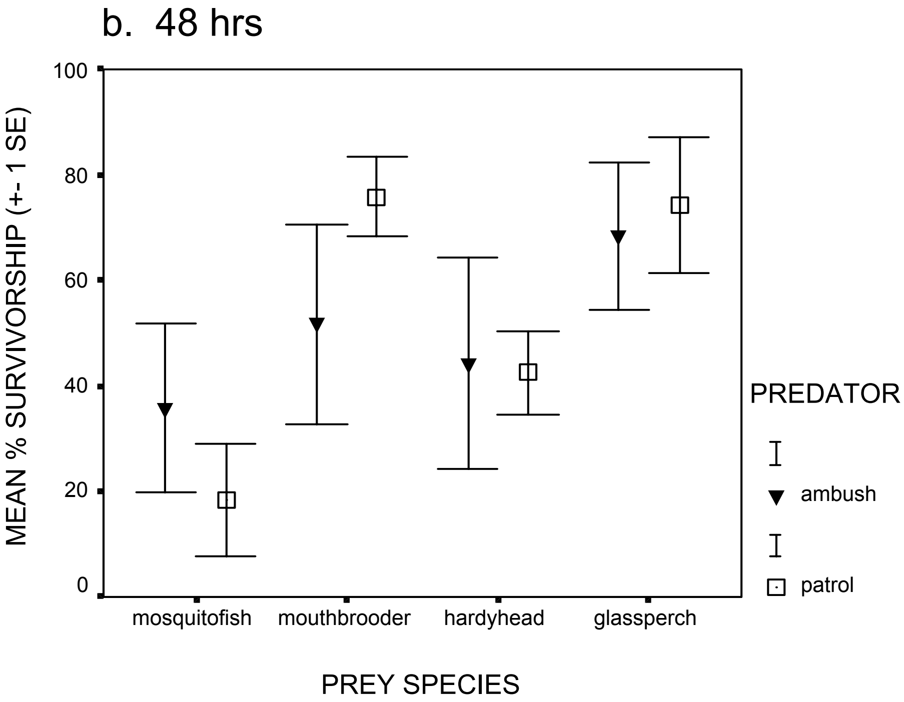


Figure 5.17 Comparison of prey species survivorship with ambush and patrol predators at different time intervals: a, 24 hrs, b, 48 hrs, c, 72 hrs

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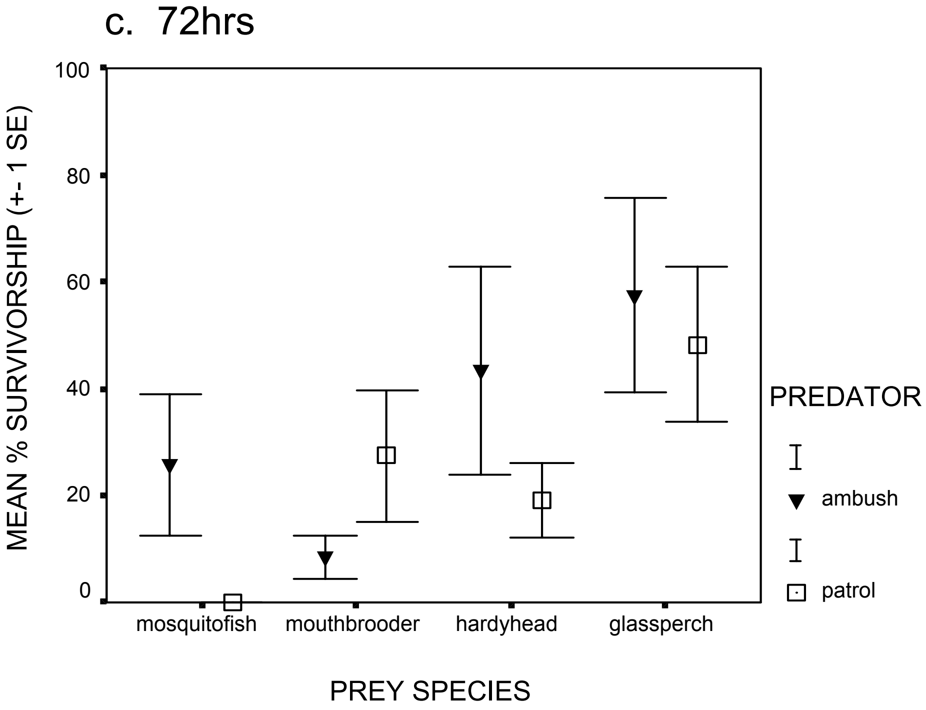


Figure 5.17 continued

Table 5.4 Statistical summary for comparison of prey class survivorship between ambush and patrol predators at different time intervals (n = 12)

Hrs	PREY CLASS	AMBUSH predators % survivorship		PATROL predators % survivorship		ANOVA	
		mean	se	mean	se	F	p
24	Slow	66.7	10.41	75.4	6.92	0.709	0.409
	Fast	82.9	5.42	85.8	3.68	0.000	0.986
	Deep body	89.2	3.47	90.8	3.30	0.000	0.994
	Narrow body	60.4	10.01	70.4	6.04	0.884	0.357
48	Slow	43.7	12.09	47.1	10.69	0.112	0.741
	Fast	56.2	12.17	58.3	8.67	0.244	0.626
	Deep body	60.0	11.50	75.0	7.12	1.664	0.210
	Narrow body	40.0	12.29	30.4	7.35	0.076	0.786
72	Slow	8.7	3.08	13.7	7.21	0.007	0.934
	Fast	50.4	12.92	33.7	8.86	0.234	0.633
	Deep body	32.9	11.62	37.9	9.62	0.221	0.643
	Narrow body	26.2	0.87	9.58	4.41	1.286	0.269

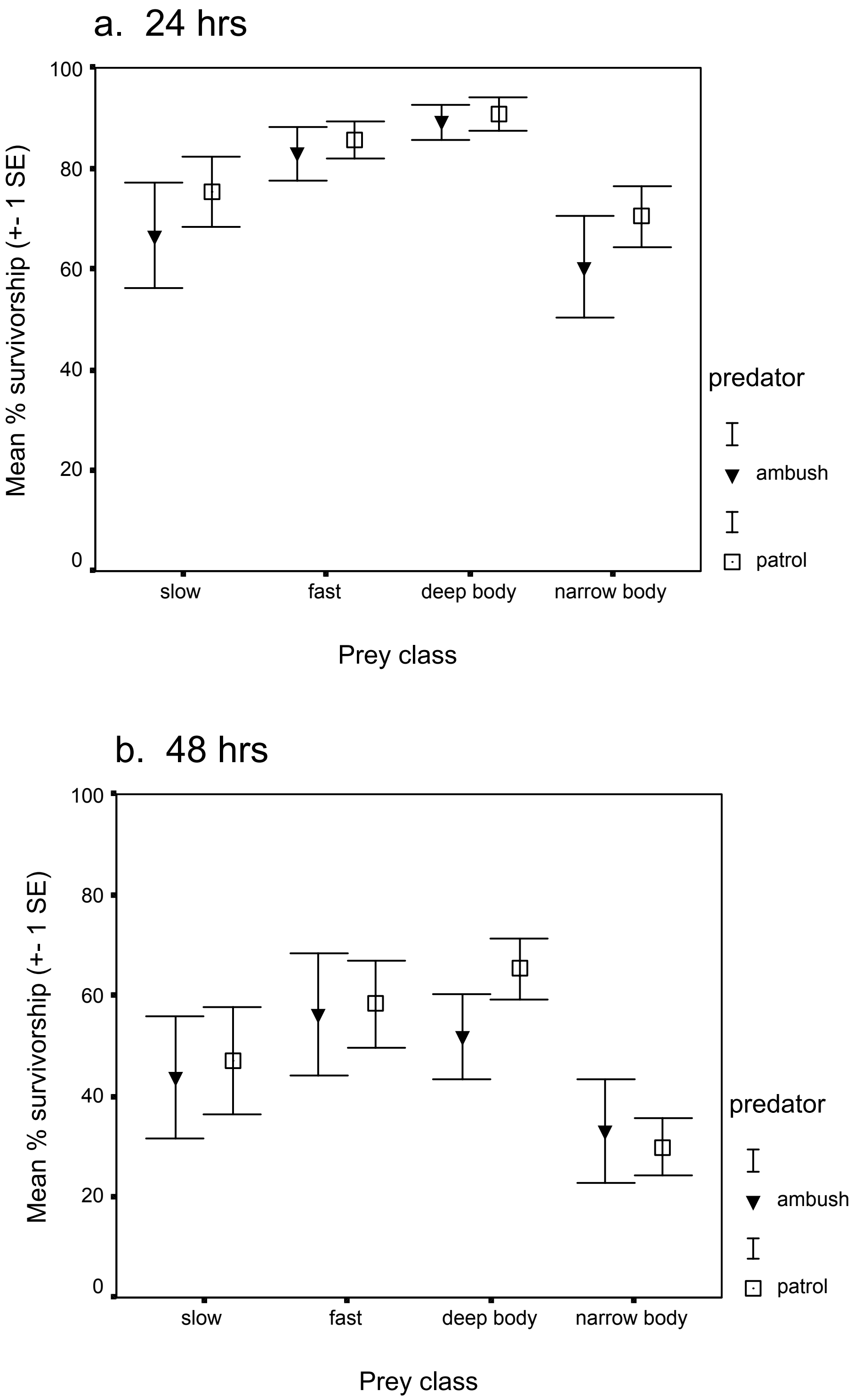


Figure 5.18 Comparison of prey class survivorship between ambush and patrol predators at different time intervals: a, 24 hrs, b, 48 hrs, c, 72 hrs continued

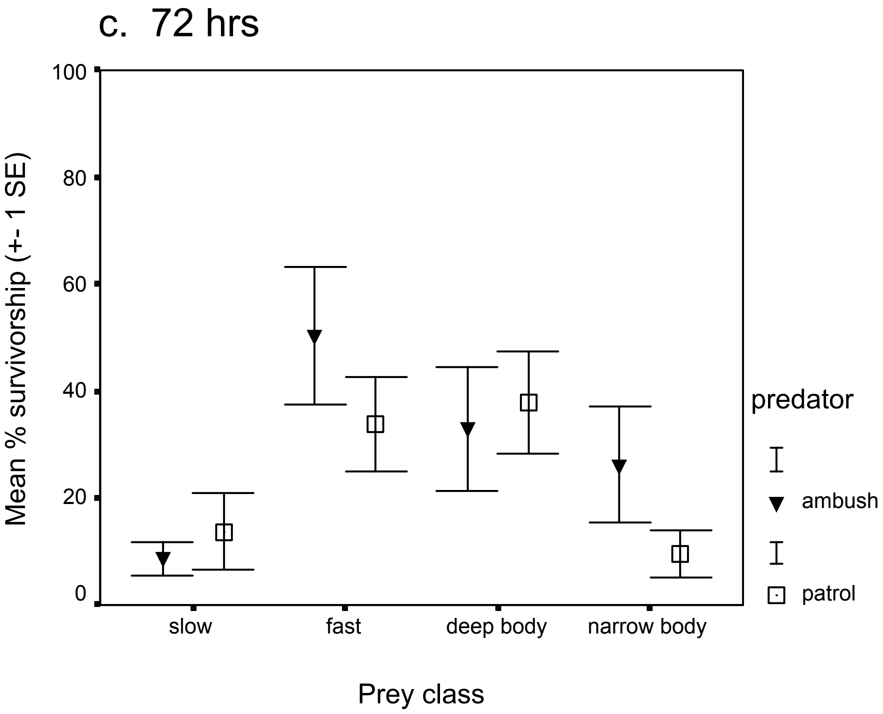


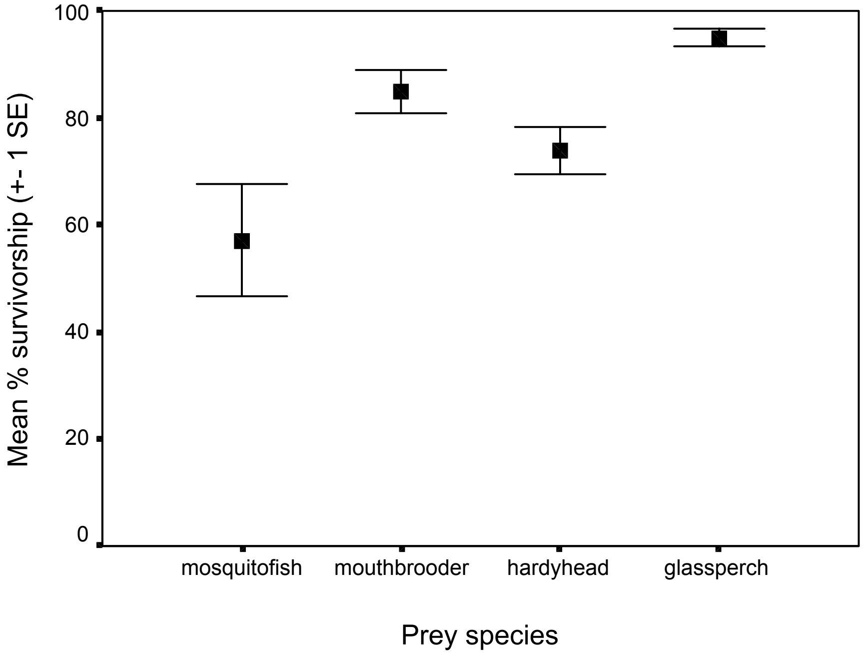
Figure 5.18 continued

At all time intervals, the non-indigenous Mosquitofish had significantly lower survivorship than the indigenous Agassiz's glassperch irrespective of predator mode, while there was no significant difference in survivorship between Mozambique mouthbrooder, Fly-specked hardyhead and Agassiz's glassperch (Table 5.5 and Figure 5.19). There was significantly lower survivorship at 72 hours of slow (non-indigenous) fish compared with fast (indigenous) fish, and significantly lower survivorship of narrow-bodied fish at 24 hours and 48 hours compared with deep-bodied fish (Table 5.6 and Figure 5.20).

Table 5.5 Statistical summary for comparison of prey species and classes survivorship in the presence of a predator at different time intervals (GAM = Mosquitofish, GP = Agassiz's glassperch, sample size: prey species, n = 12, prey classes, n = 24)

Hrs	**PREY SPECIES**											
	Mosquitofish		**Mouthbrooder**		**Hardyhead**		**Glassperch**		**ANOVA**		**TUKEY TEST**	
	% survivorship		% survivorship		% survivorship		% survivorship		F	p		
	mean	se	mean	se	mean	se	mean	se				
24	50.6	8.29	70.8	3.98	60.7	3.58	80.0	2.74	6.154	0.001	Gam ≠ GP	
48	24.7	7.57	54.8	8.26	38.3	8.54	62.3	6.84	4.657	0.007	Gam ≠ GP	
72	0.1	0.06	0.3	0.08	0.4	0.11	0.6	0.10	5.737	0.010	Gam ≠ GP	
	PREY CLASSES											
	SLOW		FAST		ANOVA		DEEP		NARROW		ANOVA	
	% survivorship		% survivorship				% survivorship		% survivorship		F	p
	mean	se	mean	se			mean	se	mean	se		
24	60.7	4.96	70.4	2.98	2.801	0.101	75.4	2.55	55.6	4.54	14.371	<0.001
48	39.7	6.32	50.3	5.91	1.492	0.228	58.6	5.30	31.5	5.76	12.006	0.001
72	0.2	0.05	0.5	0.08	10.649	0.002*	0.5	0.07	0.2	0.07	4.719	0.035

a. 24 hrs



b. 48 hrs

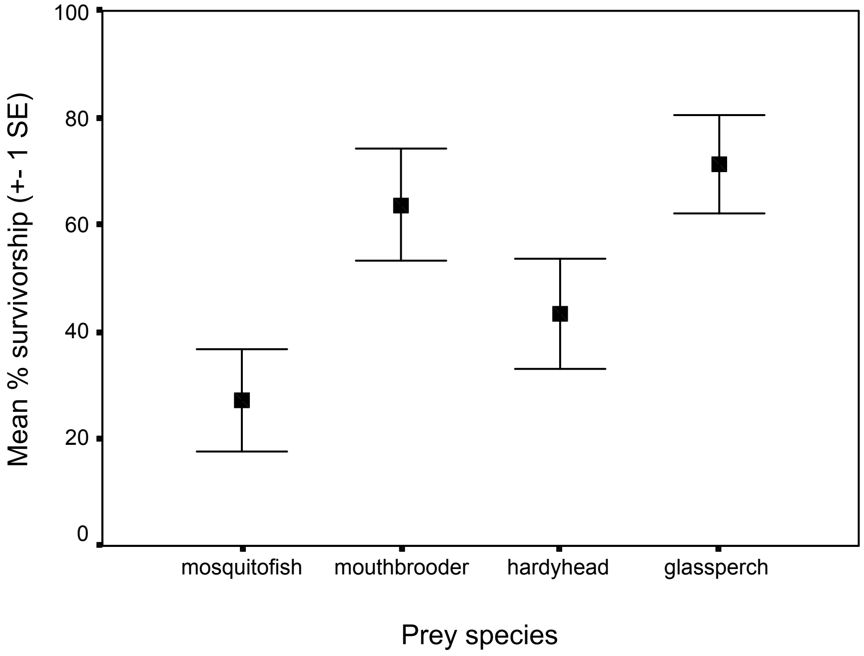


Figure 5.19 Prey species survivorship at different time intervals in the presence of a predator: a, 24 hrs, b, 48hrs, c, 72 hrs continued

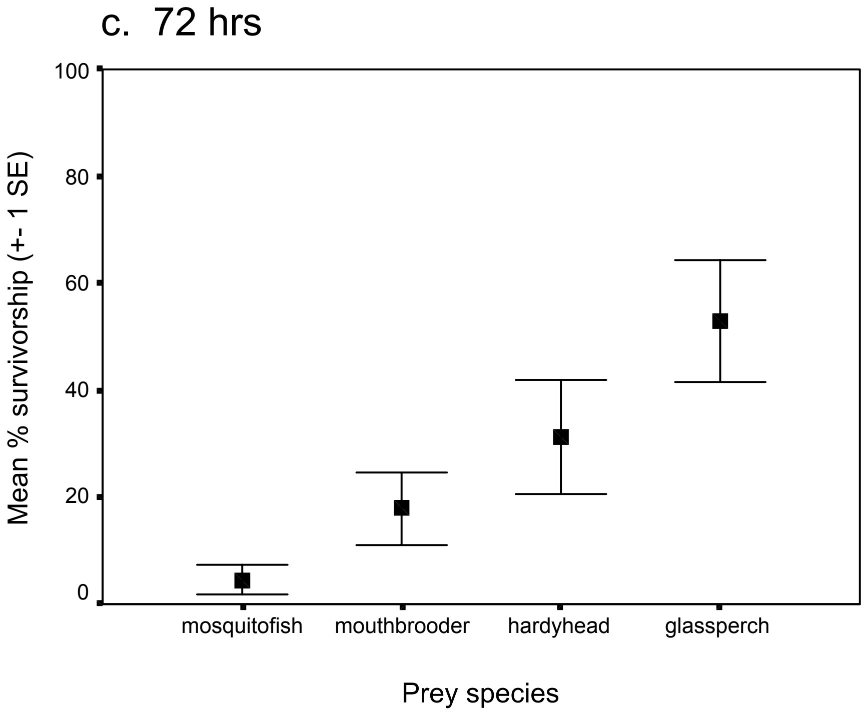
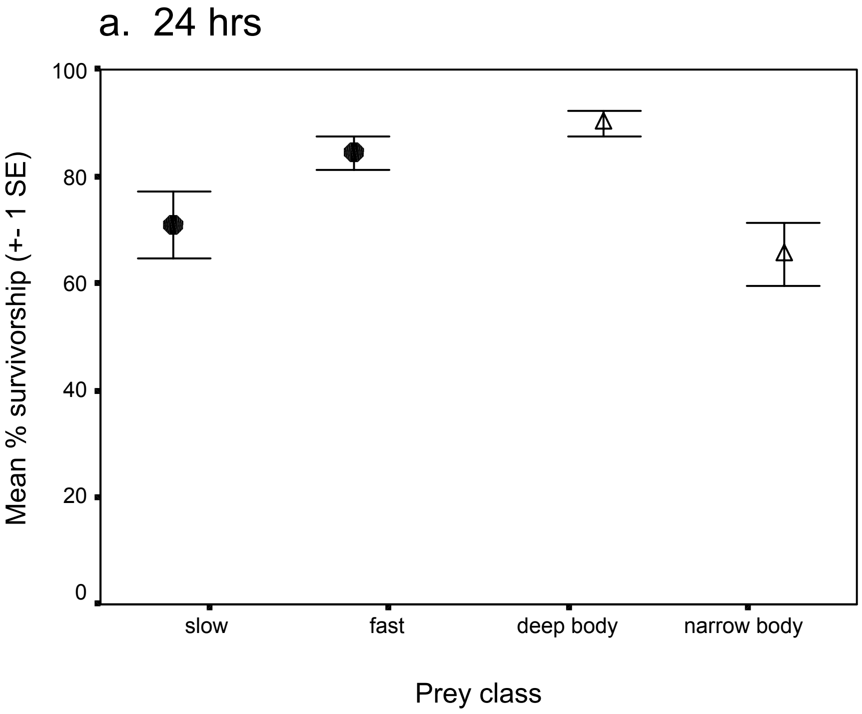


Figure 5.19 continued



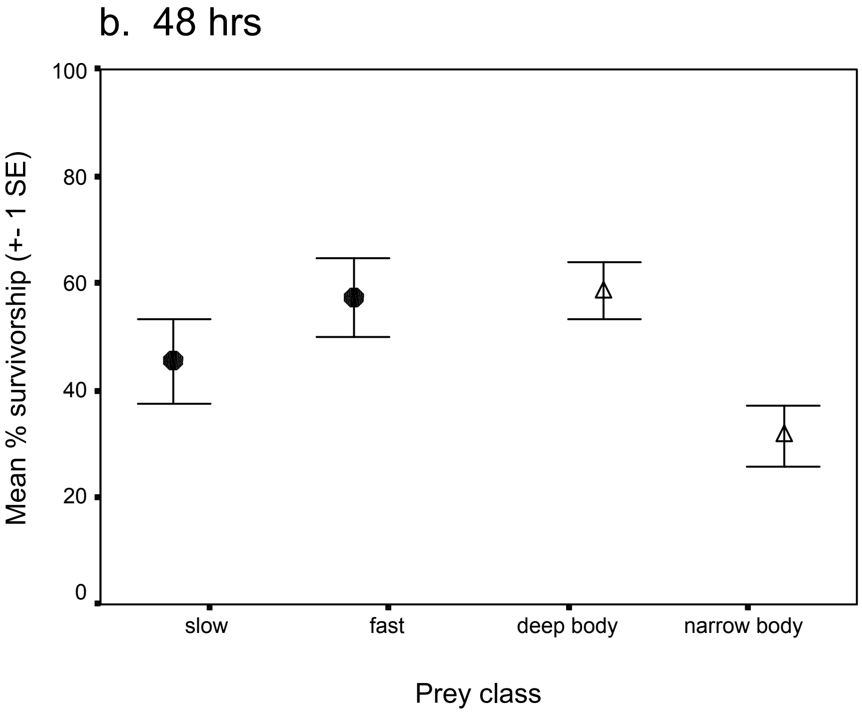


Figure 5.20 Prey class (Speed, Morphology) survivorship in the presence of a predator at different time intervals: a, 24 hrs, b, 48hrs, c, 72 hrs continued

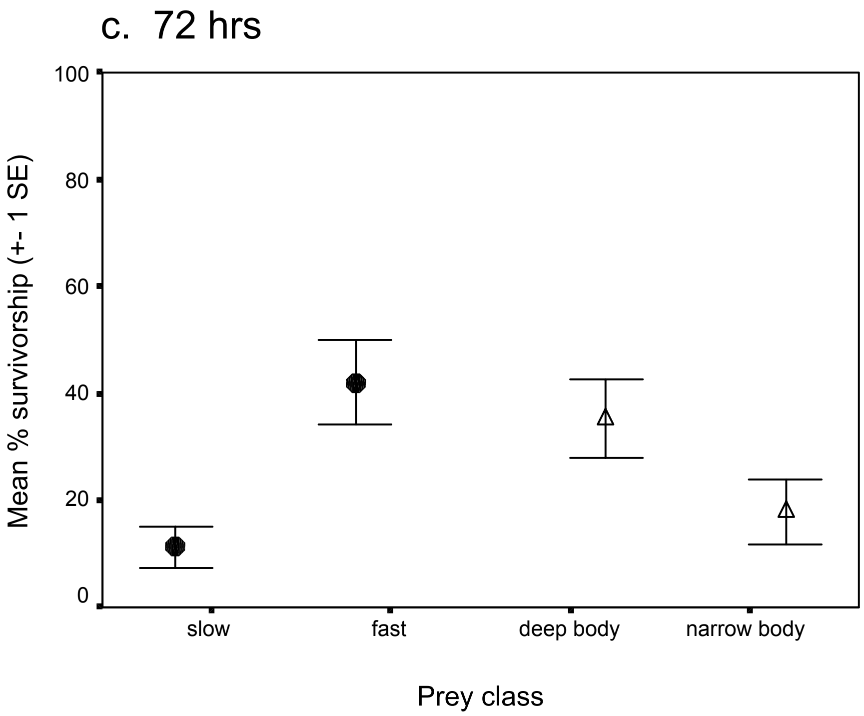


Figure 5.20 continued

5.3.3 Experiment 3: Survivorship of non-indigenous fish species in mixed prey populations at different densities in the presence of different predator classes

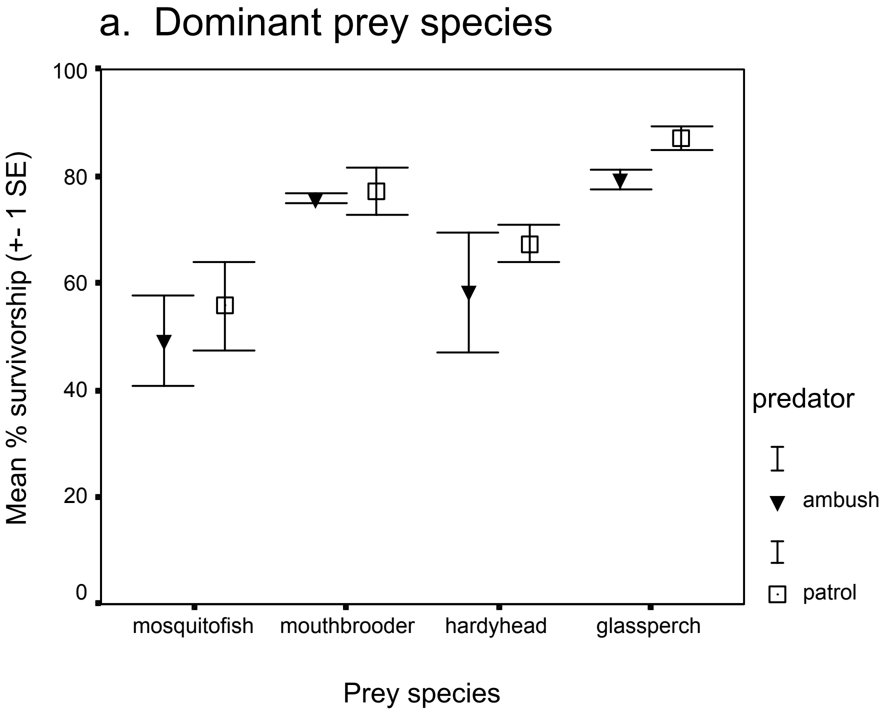
Survivorship of "dominant" prey species and classes

The results for comparison of mean survivorship of dominant prey species and dominant prey classes between predator classes are presented in Table 5.6 and Figure 5.21. There was no difference in survivorship for all prey species and all prey classes between ambush and patrol predators. Mean survivorship ranking (low to high) for prey species and prey classes was the same for both types of predator:

SPECIES: Mosquitofish < Hardyhead < Mozambique mouthbrooder < Glassperch
CLASS: narrow body < slow < fast < deep body

Table 5.6 Statistical summary for comparison of survivorship of dominant prey species and classes between ambush and patrol predators

SPECIES/CLASS	**PREDATOR**				**ANOVA**	
	AMBUSH		**PATROL**			
	% survivorship					
	mean	se	mean	se	F	p
Mosquitofish	49.2	8.40	55.8	8.29	0.347	0.569
Mouthbrooder	75.9	0.98	77.2	4.47	0.185	0.677
Hardyhead	58.3	11.31	67.4	3.61	0.452	0.517
Glassperch	79.4	1.95	87.2	2.15	7.120	0.024
SLOW	62.6	5.69	66.5	5.53	0.328	0.572
FAST	68.9	6.33	77.3	3.59	1.413	0.247
DEEP BODY	77.7	1.17	82.2	2.80	2.817	0.107
NARROW BODY	53.8	6.56	61.6	4.65	0.815	0.377



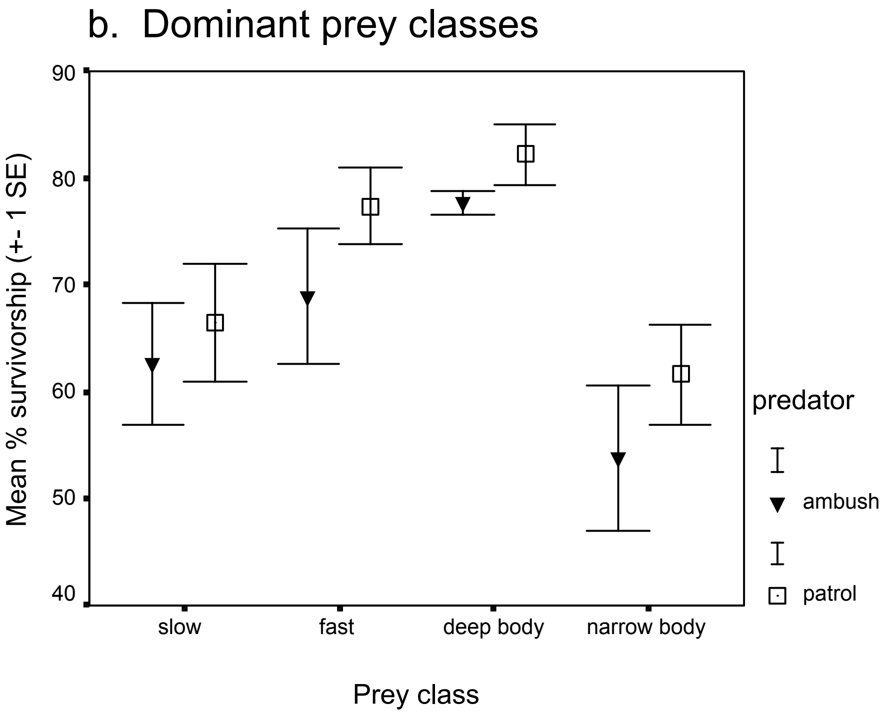


Figure 5.21 Comparison of prey survivorship (%) between ambush and patrol predators: a, dominant prey species, b, dominant prey classes

Survivorship of non-dominant prey species and classes between ambush and patrol predators

The results are presented in Table 5.7 and Figure 5.22. There was significantly lower survivorship of Mosquitofish and Fly-specked hardyhead, fast fish and narrow-bodied fish with patrol predators compared with ambush predators. Mean survivorship rankings for prey species and classes were similar for ambush and patrol predators:

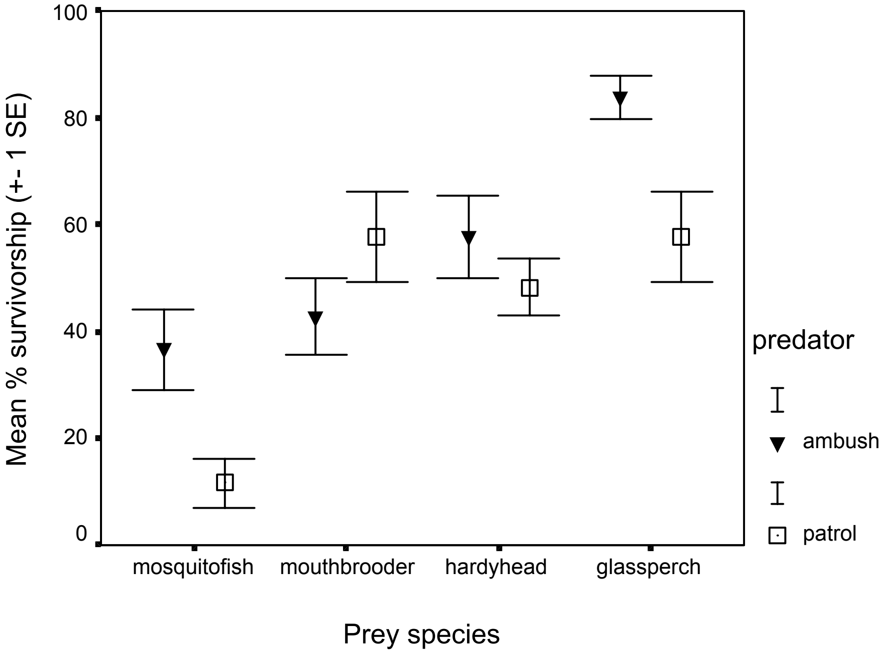
SPECIES (ambush): Mosquitofish < Mouthbrooder < Hardyhead < Glassperch
(patrol): Mosquitofish < Hardyhead < Mouthbrooder = Glassperch
CLASSES (ambush): slow < narrow < deep < fast
(patrol): narrow < slow < fast < deep

Mean survivorship rankings were similar to those for dominant prey species and classes: Mosquitofish had the lowest survivorship and Agassiz's glassperch the highest survivorship and slow/narrow-bodied fishes had lower survivorship than fast/deep-bodied fishes for both ambush and patrol predators.

Table 5.7 Statistical summary for comparison of survivorship of non-dominant prey species and classes between ambush and patrol predators

SPECIES/CLASS	**PREDATOR**				**ANOVA**	
	AMBUSH		**PATROL**			
	% survivorship					
	mean	Se	mean	se	F	p
Mosquitofish	42.8	8.05	11.7	4.52	10.415	0.003
Mouthbrooder	47.8	7.74	57.8	8.42	0.783	0.383
Hardyhead	67.2	5.23	48.3	5.25	6.720	0.014
Glassperch	78.3	6.58	57.8	8.50	2.998	0.092
SLOW	45.3	5.53	34.72	6.11	1.453	0.232
FAST	72.8	4.25	53.05	4.99	7.706	0.007
DEEP BODY	63.1	5.63	57.8	5.90	0.242	0.624
NARROW BODY	55.0	5.17	30.0	4.61	11.798	0.001

a. Non-dominant prey species



b. Non-dominant prey class

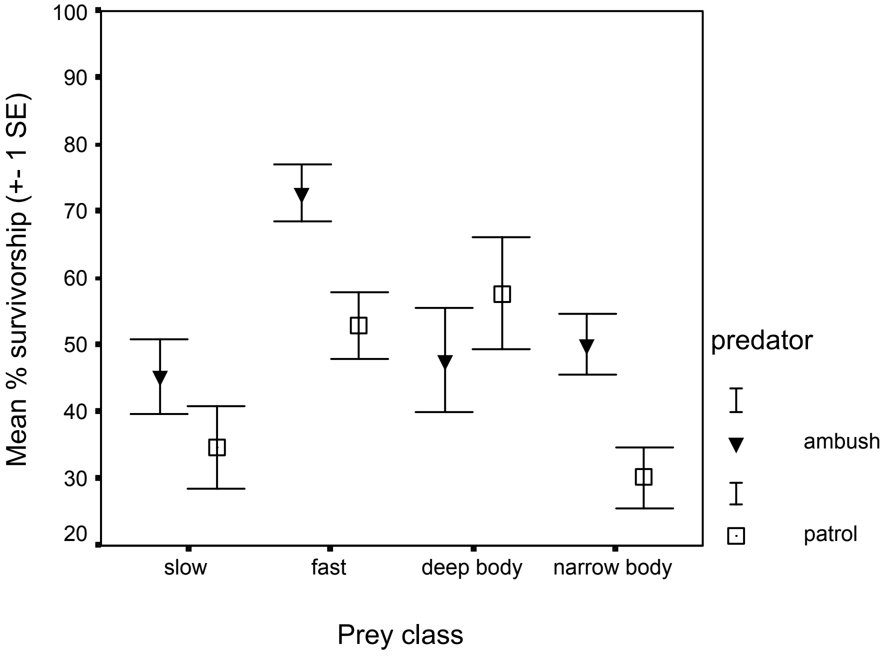


Figure 5.22 Comparison of prey survivorship (%) between ambush and patrol predators: a, non-dominant prey species, b, non-dominant prey classes

A statistical summary of comparisons of dominant prey species and class survivorship in the presence of a predator irrespective of foraging mode is presented in Table 5.8 and Figure 5.23. There were significant differences in survivorship among dominant prey species and between prey classes. Mosquitofish had significantly lower survivorship than the Mozambique mouthbrooder and Agassiz's glassperch and the glassperch had significantly higher survivorship than the Fly-specked hardyhead. Narrow-bodied fish had significantly lower survivorship than deep-bodied fish.

The mean survivorship rankings were:

SPECIES: Mosquitofish < Hardyhead < Mouthbrooder < Glassperch
CLASS: Narrow-body < Slow < Fast < Deep-body

Table 5.8 Statistical summary for comparison of dominant prey species and class survivorship in the presence of a predator

Dominant Prey Survivorship (%)

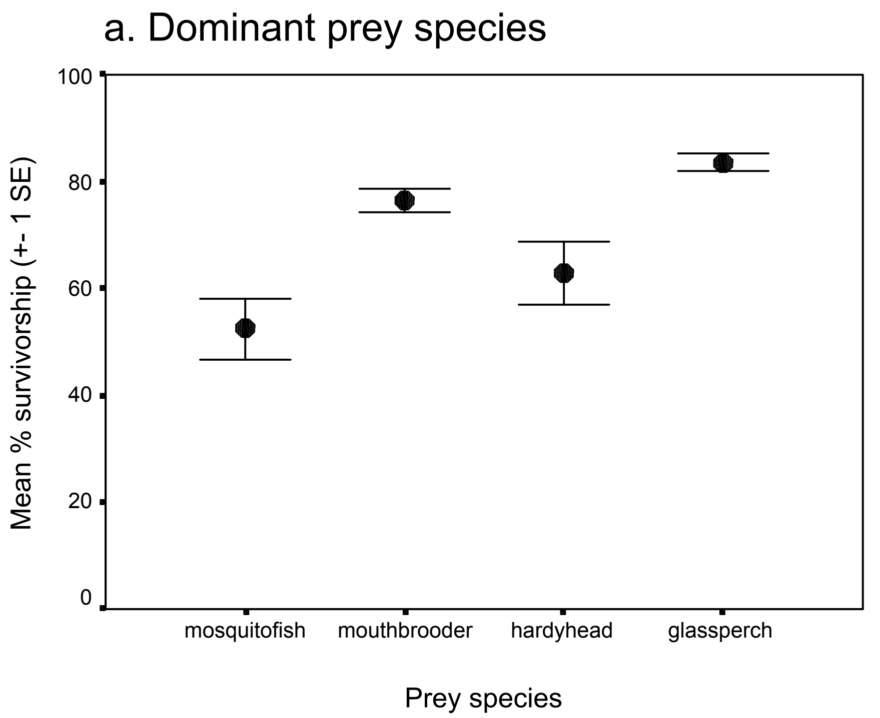
SPECIES	(n)	mean	se	**CLASS**	(n)	mean	se
Mosquitofish	12	52.5	5.71	Slow	24	64.5	3.90
Mouthbrooder	12	78.6	2.19	Fast	24	73.2	3.66
Hardyhead	12	62.9	5.82	Narrow	24	57.7	4.13
Glassperch	12	83.5	1.81	Deep	24	78.3	1.53

ANOVA

	SPECIES (p:0.01)	**CLASS (SPEED)** (p: 0.01)	**CLASS (MORPHOLOGY)** (p: 0.01)
F	10.385	2.712	25.608
p	<0.001	0.106	<0.001

TUKEY HSD - differences
Mosquitofish ≠ Mouthbrooder (p = 0.002)
Mosquitofish ≠ Glassperch (p = <0.001
Hardyhead ≠ Glassperch (p = 0.007)

A statistical summary of non-dominant prey and class survivorship in the presence of a predator irrespective of foraging mode is presented in Table 5.9 and Figure 5.24. There were significant differences in survivorship among non-dominant prey species and between prey classes. Mosquitofish had significantly lower survivorship than the other three prey species. Slow fish had significantly lower survivorship than fast fish and narrow-bodied fish had significantly lower survivorship than deep-bodied fish.



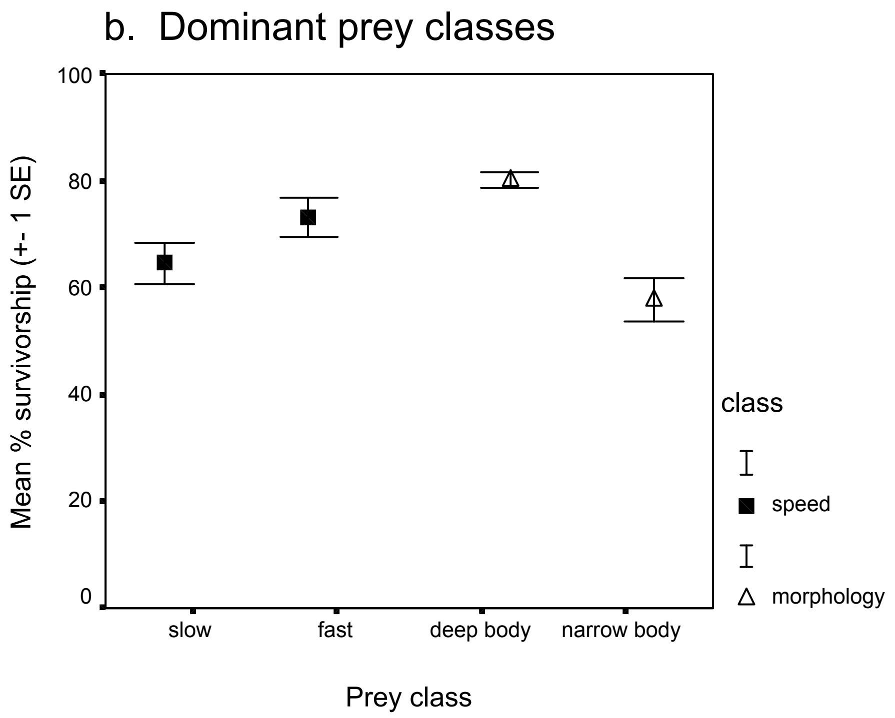


Figure 5.23 Comparison of dominant prey survivorship (%) in the presence of a predator: a, prey species, b, prey classes

The mean survivorship rankings were:

SPECIES: Mosquitofish < Mouthbrooder < Hardyhead < Glassperch
CLASS: Slow < Narrow-body < Deep-body < Fast

Table 5.9 Statistical summary for non-dominant prey species and class survivorship in the presence of a predator

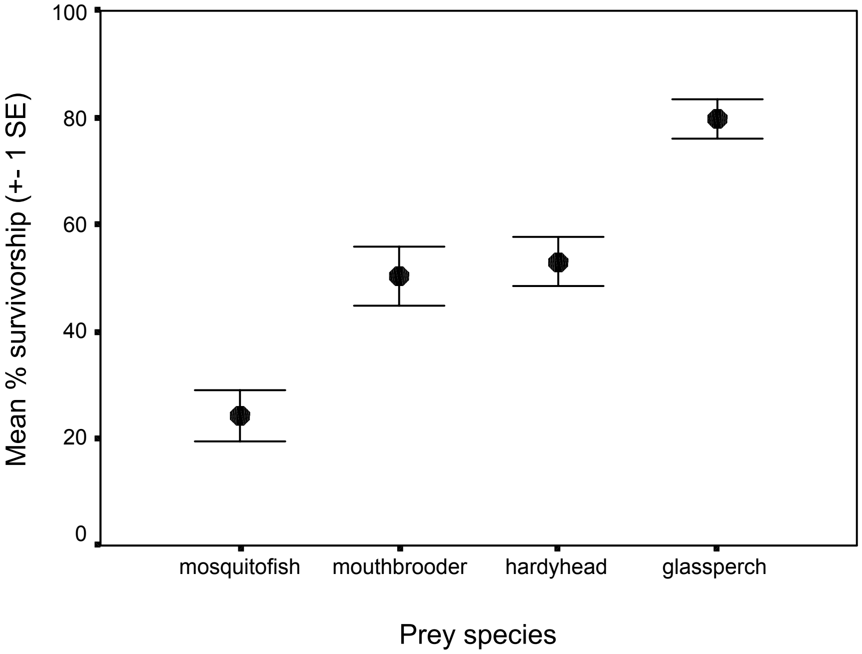
		Prey Survivorship (%)					
SPECIES	**(n)**	**mean**	**se**	**CLASS**	**(n)**	**mean**	**se**
Mosquitofish	36	27.2	5.26	Slow	72	40.0	4.14
Mouthbrooder	36	52.8	5.70	Fast	72	62.9	3.46
Hardyhead	36	57.8	3.99	Narrow	72	42.5	3.75
Glassperch	36	68.1	5.57			60.4	4.06

		ANOVA	
	SPECIES	**CLASS (SPEED)**	**CLASS (MORPHOLOGY)**
F	12.442	20.095	11.315
P	<0.001	<0.001	0.001

TUKEY HSD – differences
Mosquitofish ≠ Mooouthbrooder (p = 0.002)
Mosquitofish ≠ Hardyhead (p = <0.001)
Mosquitofish ≠ Glassperch (p = <0.001)

Dominant and non-dominant prey species and classes in the presence of ambush or patrol predators or irrespective of predator mode had similar survivorship rankings with Mosquitofish having the lowest survivorship and Agassiz’s glassperch the highest survivorship and with slow/narrow-bodied fish with lower survivorship than fast/deep-bodied fish.

a. Non-dominant prey species



b. Non-dominant prey class

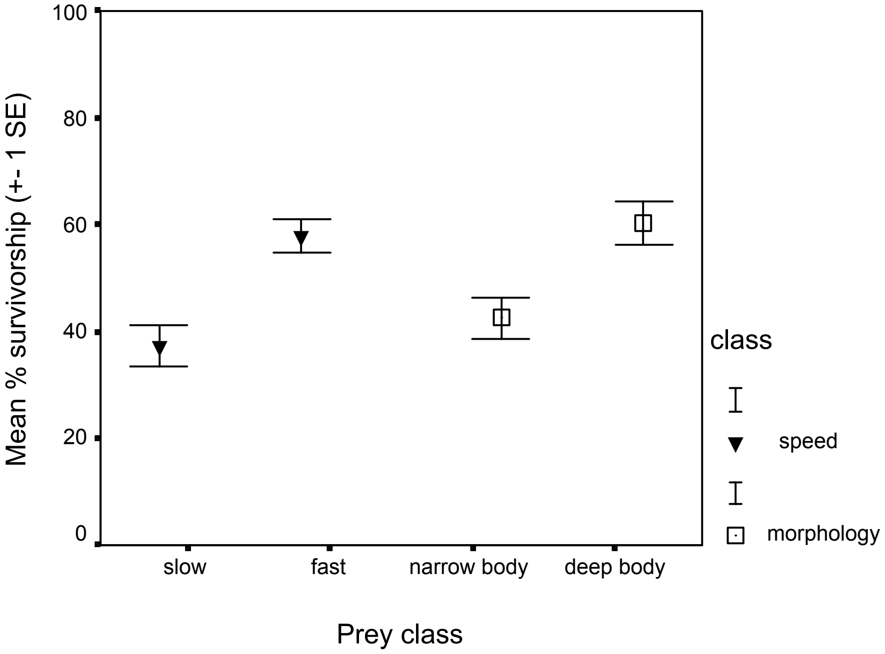


Figure 5.24 Comparison of non-dominant prey survivorship (%) in the presence of a predator: a, prey species, b, prey classes

5.3.4 Experiment 4: Survivorship of non-indigenous fish species in mixed prey populations in relation to vegetation cover in the presence of different predator classes

Comparison of prey species and prey class survivorship between ambush and patrol predators

There were significant differences in mean survivorship of prey species between ambush and patrol predators, within and among levels of cover, and with significant interaction effects (Table 5.10). At zero cover, there was no significant difference in prey species survivorship between predator classes. At medium cover, survivorship of Mosquitofish and Mozambique mouthbrooder was significantly lower with ambush predators than with patrol predators. At high cover, survivorship of Mozambique mouthbrooder was significantly lower with ambush predators than with patrol predators (Figure 5.25).

Survivorship for all prey species increased with cover in the presence of either predator type. At zero cover the non-indigenous Mosquitofish had the lowest survivorship with patrol predators and the non-indigenous Mozambique mouthbrooder had the lowest survivorship with ambush predators (Figure 5.23), but both species had the highest survivorship at medium and high cover with patrol predators (Figure 5.23) in comparison with survivorship of indigenous prey species. Both the Mosquitofish and Mozambique mouthbrooder, which were both relatively pale in colour in open water, became much darker in vegetation and the vertical barring of Mozambique mouthbrooder juveniles became more pronounced. No colour changes were observed in either indigenous fish species in open water or with vegetation cover present. At zero cover, Agassiz's glassperch had the highest survivorship with ambush and patrol predators and Fly-specked hardyhead had the highest survivorship with ambush predators compared with non-indigenous fishes. At high cover, Fly-specked hardyhead had the lowest survivorship of all prey species with ambush and patrol predators (Figure 5.25).

There were significant differences in prey class survivorship between ambush and patrol predators at different levels of cover (Table 5.10). At zero cover, there was no significant difference in survivorship between fast fish and slow fish between predator classes, and between narrow-bodied fish and deep-bodied fish between predator classes (Figure 5.24a). Slow fish and deep-bodied fish had significantly lower survivorship with ambush predators compared with patrol predators at medium and high levels of cover (Figure 5.26).

Table 5.10 Statistical summary for comparison of prey species and class survivorship between ambush and patrol predators at different levels of cover

ANOVA	df	F	p
COVER	2	51.176	<0.001
SPECIES	3	5.176	0.002
PREDATOR	1	5.895	0.010
COVER * SPECIES	6	6.348	<0.001
COVER * PREDATOR	2	6.819	0.002
SPECIES * PREDATOR	3	9.207	<0.001
COVER * SPECIES * PREDATOR	6	0.738	0.620

COVER	PREY SPP/CLASS	PREDATOR CLASS				ANOVA	
		AMBUSH		PATROL			
		% survivorship		% survivorship			
		mean	se	mean	se	F	p
NONE	Mosquitofish	9.2	5.07	0.0	0.00	4.078	0.071
	Mouthbrooder	8.3	3.33	26.7	12.49	1.215	0.296
	Hardyhead	43.3	19.43	19.2	7.00	0.589	0.461
	Glassperch	60.0	16.88	48.3	14.53	0.282	0.607
MEDIUM	Mosquitofish	51.7	6.01	84.2	7.46	10.347	0.009
	Mouthbrooder	15.8	3.00	95.0	2.24	164.156	<0.001
	Hardyhead	45.8	18.73	34.2	9.87	0.206	0.660
	Glassperch	55.0	14.55	68.3	7.38	0.509	0.492
HIGH	Mosquitofish	78.3	6.79	92.5	1.71	4.453	0.061
	Mouthbrooder	65.0	5.32	97.5	1.12	39.268	<0.001
	Hardyhead	61.7	11.15	50.8	5.39	0.903	0.364
	Glassperch	79.17	3.27	77.5	5.59	0.030	0.866

COVER	PREY SPP/CLASS	PREDATOR CLASS				ANOVA	
		AMBUSH		PATROL			
		% survivorship		% survivorship			
		mean	se	mean	se	F	p
NONE							
	SLOW	8.8	2.87	13.3	7.19	0.000	0.996
	FAST	51.7	12.53	33.7	8.86	1.164	0.292
	NARROW	26.3	10.85	9.6	4.41	1.573	0.222
	DEEP	34.2	11.31	37.5	9.70	0.050	0.826
MEDIUM							
	SLOW	33.7	6.28	89.6	4.06	53.033	<0.001
	FAST	50.4	11.39	51.2	7.81	0.001	0.982
	NARROW	48.7	9.42	59.2	9.57	1.224	0.280
	DEEP	35.4	9.22	81.7	5.45	15.408	0.001
HIGH							
	SLOW	71.7	4.58	95.0	1.23	28.693	<0.001
	FAST	70.4	6.14	64.2	5.46	0.613	0.442
	NARROW	70.0	6.71	71.7	6.83	0.202	0.657
	DEEP	72.1	3.66	87.5	4.06	7.694	0.010

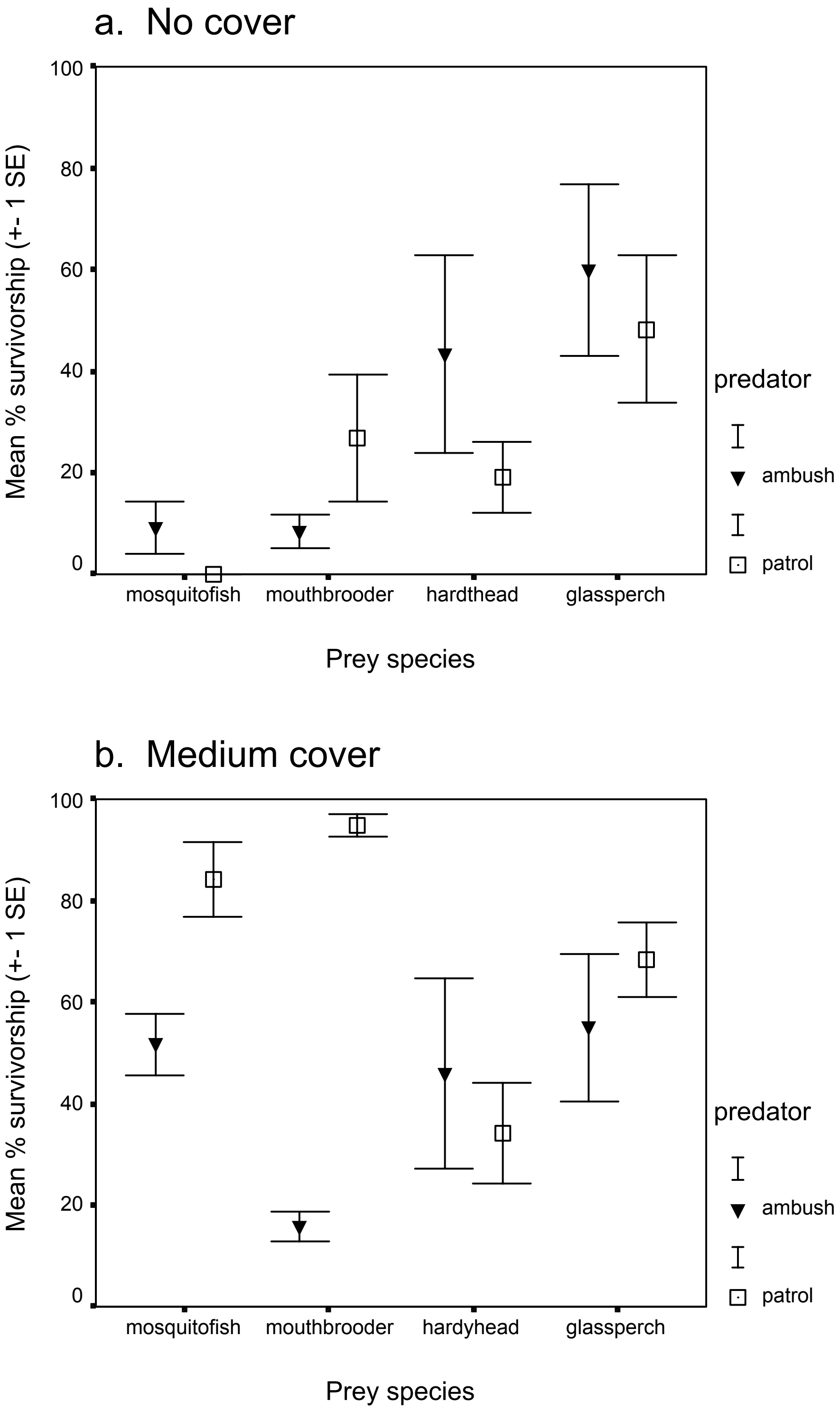


Figure 5.25 Survivorship of prey species between ambush and patrol predators at different levels of cover: a, no cover, b, medium cover, c, high cover continued

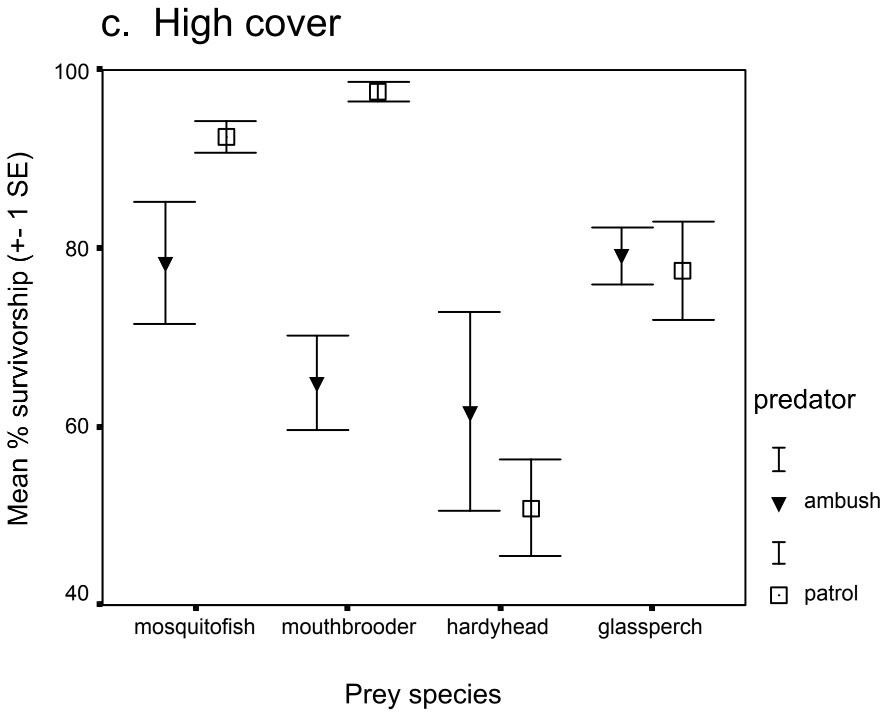
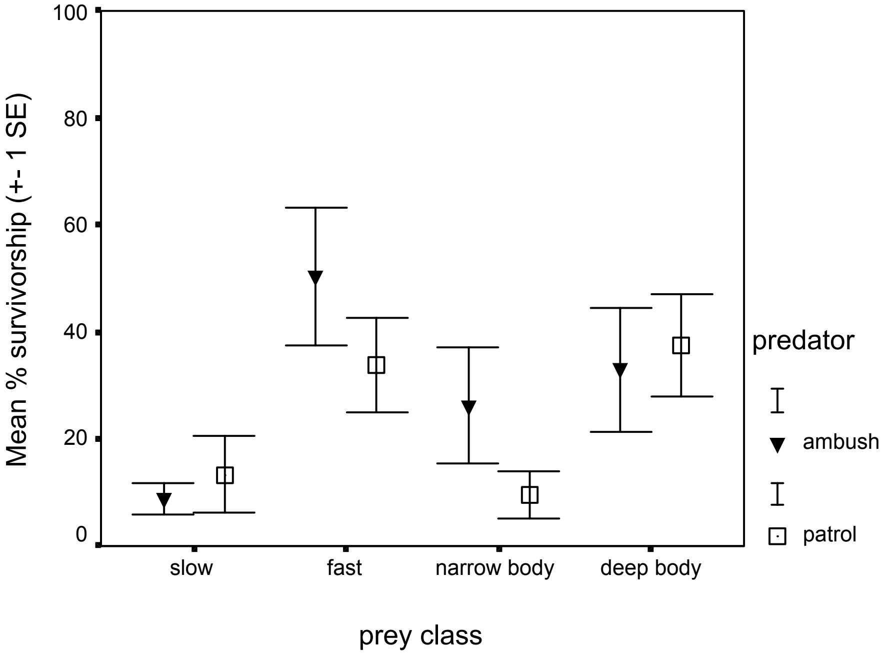


Figure 5.25 continued

a. No cover



b. Medium cover

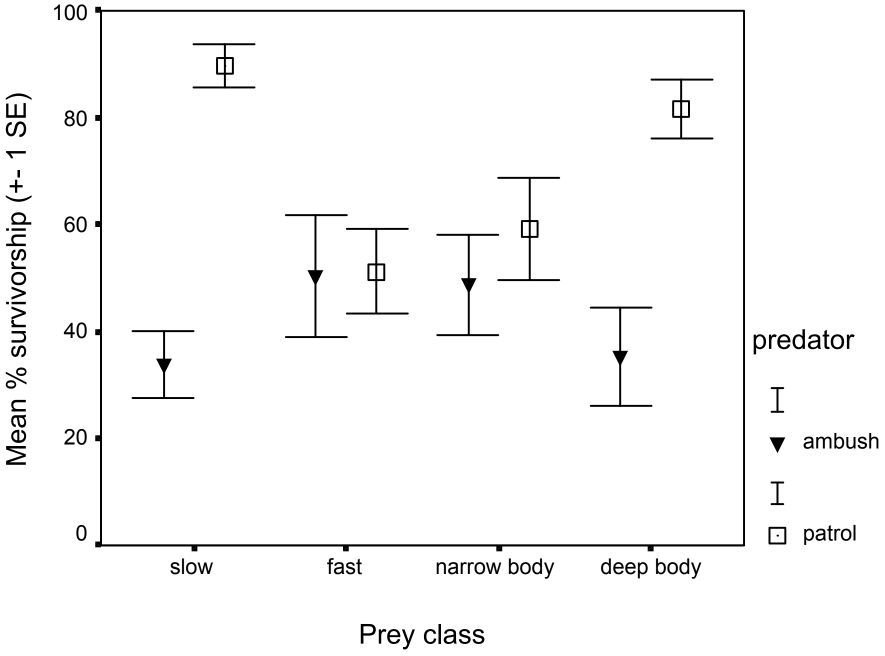


Figure 5.26 Survivorship of prey classes between ambush and patrol predators at different levels of cover: a, no cover, b, medium cover, c, high cover continued

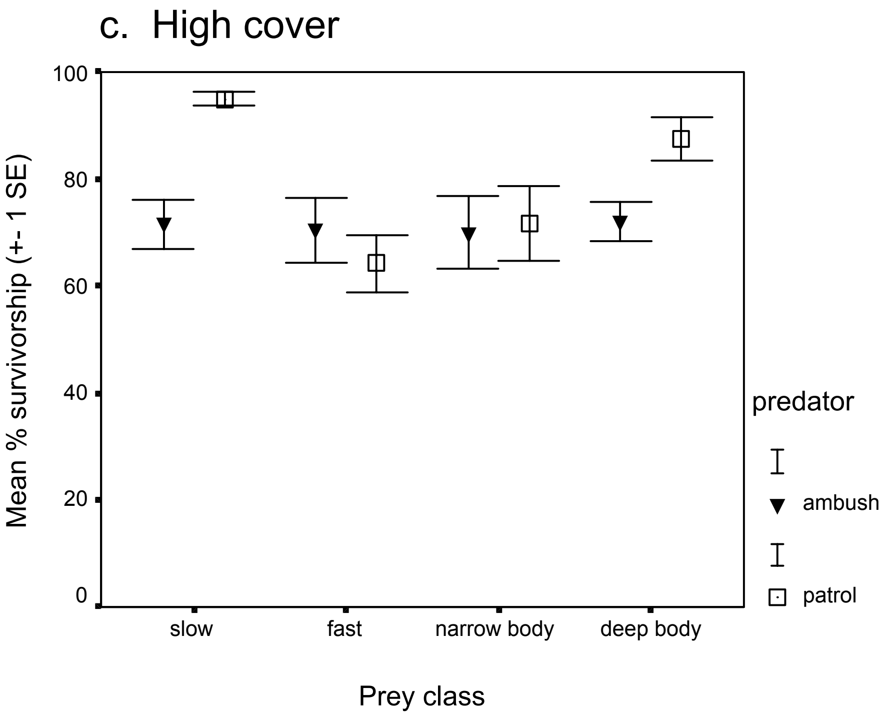


Figure 5.26 continued

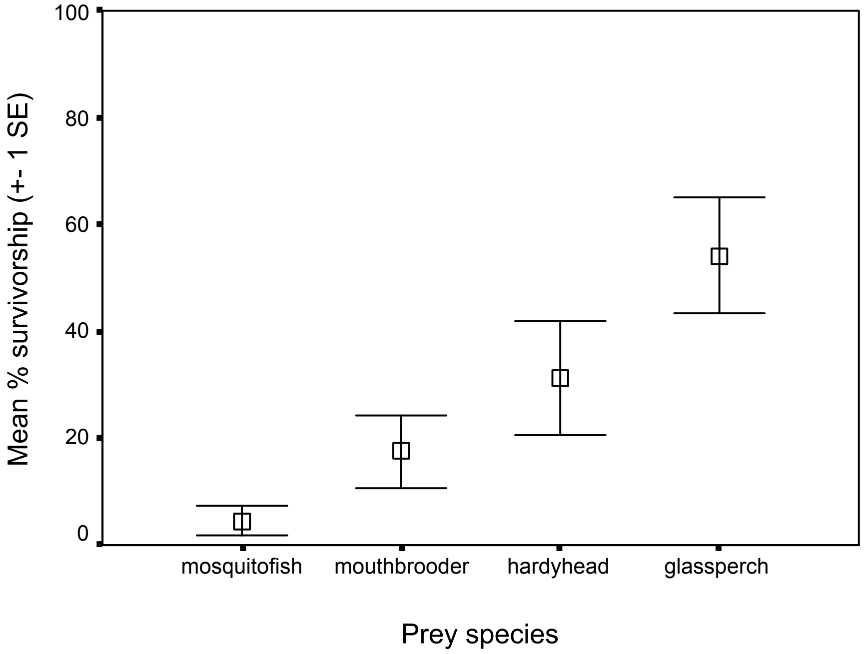
Comparison of prey species and prey class survivorship in the presence of a predator

There were significant differences in prey species survivorship irrespective of predator foraging strategy at no cover and high cover (Table 5.11). Survivorship of non-indigenous Mosquitofish was significantly lower than survivorship of indigenous Agassiz's glassperch (Figure 5.27). At high cover, survivorship of non-indigenous Mosquitofish and Mozambique mouthbrooder were significantly higher than survivorship of the indigenous Fly-specked hardyhead (Figure 5.27). At no cover, slow fish had significantly lower survivorship than fast fish, and narrow fish had significantly lower survivorship than deep-bodied fish (Figure 5.28). At medium cover there was no significant difference in survivorship between prey classes (Figure 5.28). At high cover, slow fish had significantly higher survivorship than fast fish (Figures 5.28).

Table 5.11 Statistical summary for comparison of prey species and class survivorship in the presence of a predator with different levels of cover

COVER	**PREY SPP/CLASS**	**% survivorship**		**ANOVA**		
		mean	se	F	p	Tukey test
NONE	Mosquitofish	4.6	2.78			
	Mouthbrooder	17.5	6.76			
	Hardyhead	31.2	10.50			
	Glassperch	54.2	10.76			
				7.225	0.001	Mosquitofish ≠ Glassperch
MEDIUM	Mosquitofish	67.9	6.70			
	Mouthbrooder	55.4	12.07			
	Hardyhead	40.0	10.24			
	Glassperch	61.7	8.03			
				1.576	0.209	
HIGH	Mosquitofish	85.4	3.96			
	Mouthbrooder	81.2	5.54			
	Hardyhead	56.2	6.13			
	Glassperch	78.3	3.10	6.584	0.001	Hardyhead ≠ Mosquitofish
						Hardyhead ≠ Mouthbrooder
					P (0.05)	
NONE	SLOW	11.0	3.82			
	FAST	42.7	7.73			
				13.228	0.001	
	NARROW	17.9	6.00			
	DEEP	35.8	7.30			
				5.190	0.027	
MEDIUM	SLOW	61.7	6.87			
	FAST	50.83	6.75			
				1.848	0.181	
	NARROW	54.0	6.65			
	DEEP	58.5	7.12			
				0.281	0.599	
HIGH	SLOW	83.3	3.36			
	FAST	67.3	4.07			
				11.258	0.002	
	NARROW	70.8	4.69			
	DEEP	79.8	3.12			
				2.644	0.111	

a. No cover



b. Medium cover

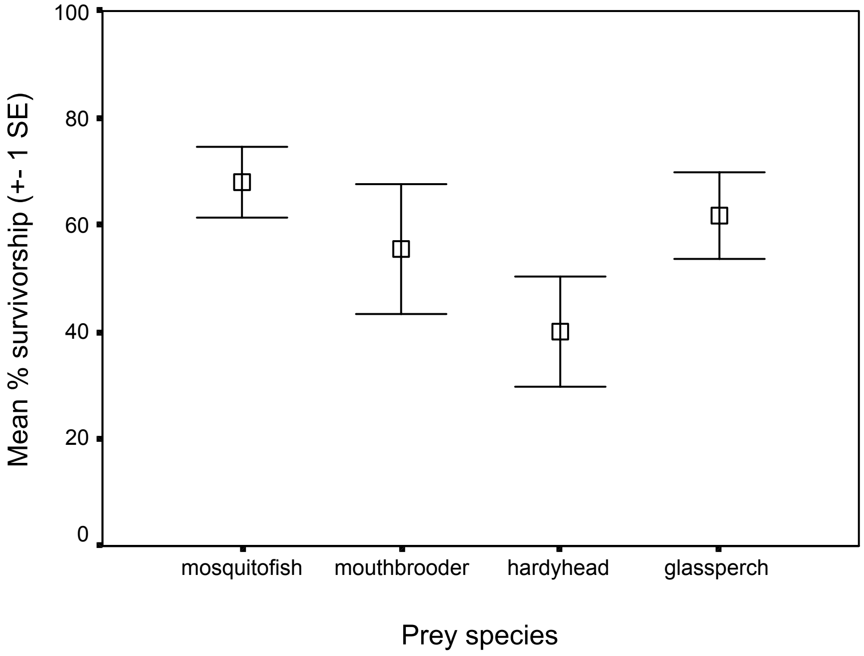


Figure 5.27 Prey species survivorship in the presence of a predator at different levels of cover: a, no cover, b, medium cover, c, high cover continued

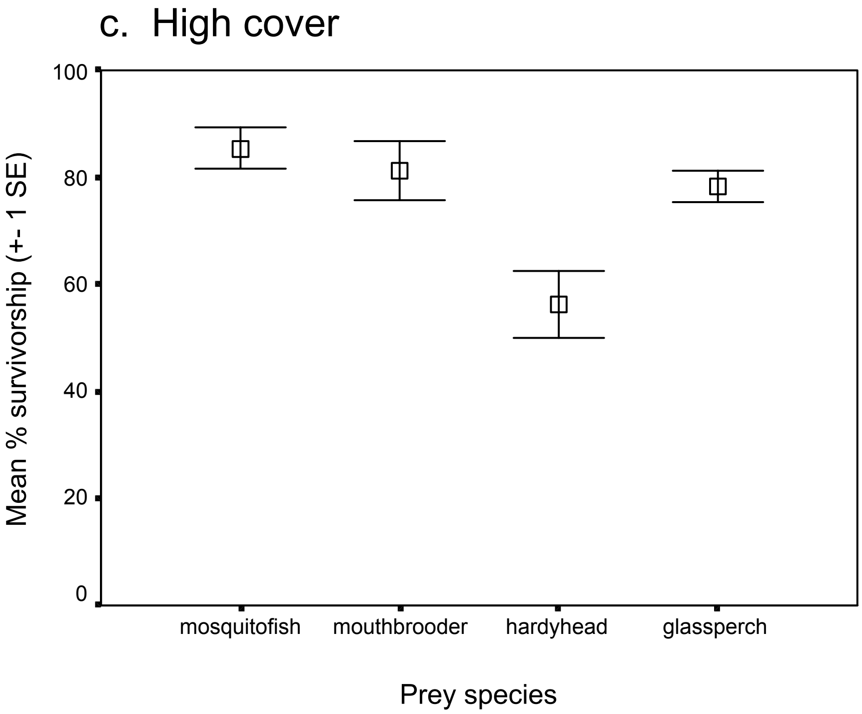
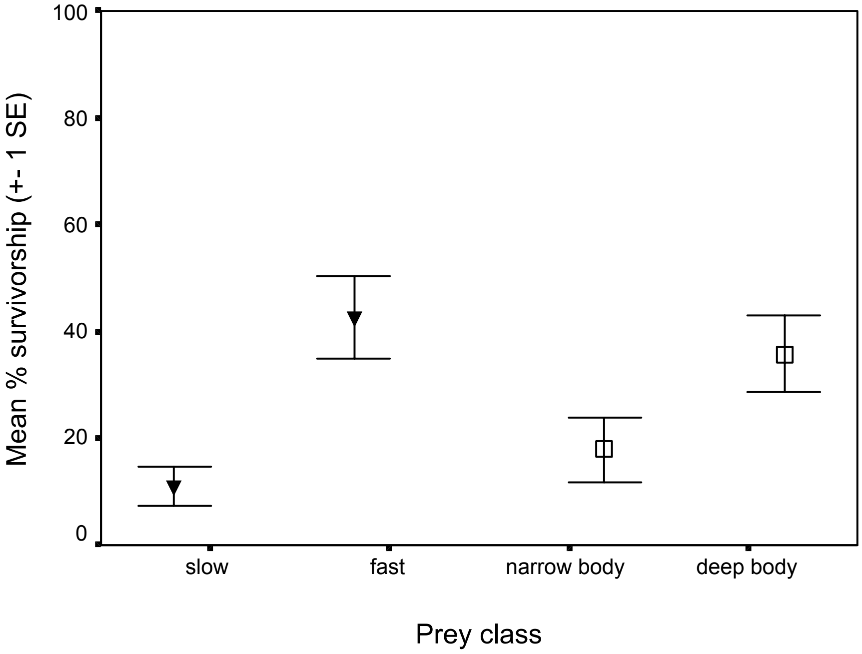


Figure 5.27 continued

a. No cover



b. Medium cover

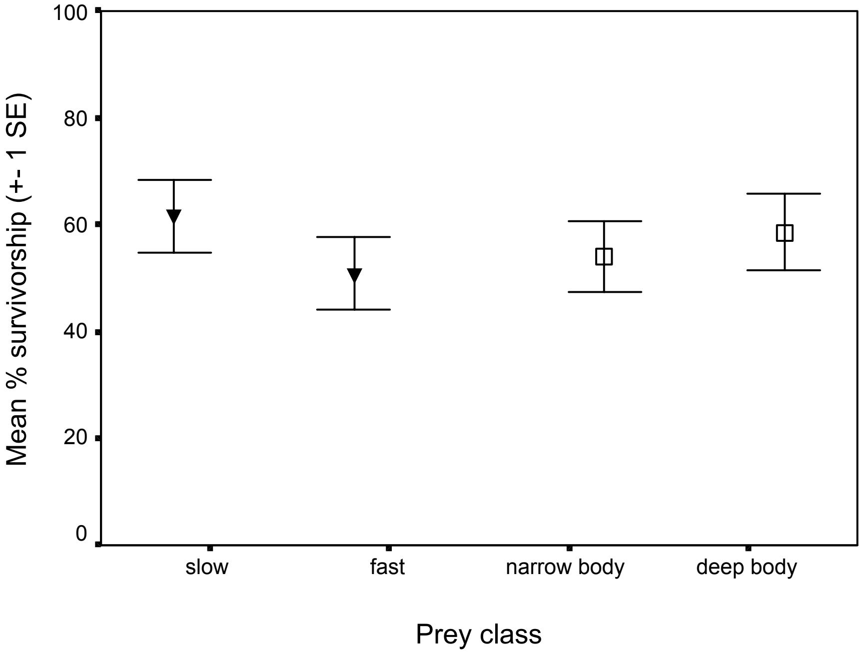


Figure 5.28 Prey class survivorship in the presence of a predator at different levels of cover: a, no cover, b, medium cover, c, high cover

continued

c. High cover

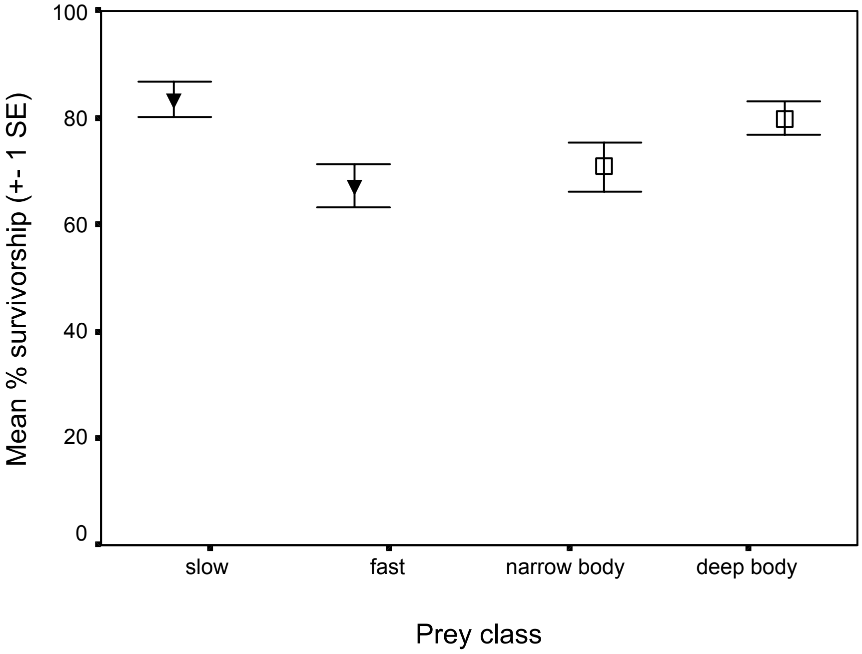


Figure 5.28 continued

Comparison of prey species and prey class survivorship between levels of cover in the presence of a predator is shown in Table 5.12. There was significantly higher survivorship for both non-indigenous prey species, the Mosquitofish and Mozambique mouthbrooder between no cover and medium and high cover. Survivorship of slow fish significantly increased between all levels of cover. Survivorship of narrow-bodied fish was significantly higher in medium cover and high cover compared with no cover. Survivorship of deep-bodied fish was significantly higher at high cover compared with no cover (refer also Figures 5.25 to 5.28).

Table 5.12 Statistical summary for comparisons of prey species and class survivorship between levels of cover in the presence of a predator

PREY SPECIES/CLASS	df	F	ANOVA p)	Tukey Test
Mosquitofish	2,33	69.669	<0.001	no cover (≠ medium cover) ≠ high cover
Mouthbrooder	2,33	14.229	<0.001	no cover (≠ medium cover), ≠ high cover
Hardyhead	2,33	2.343	0.112	
Glassperch	2,33	1.466	0.245	
SLOW	2,33	69.669	<0.001	no cover ≠ medium cover ≠ high cover
FAST	2,33	2.343	0.112	
NARROW BODY	2,33	14.229	<0.001	no cover (≠ medium cover), ≠ high cover
DEEP BODY	2,33	69.669	<0.001	no cover ≠ high cover

5.4. Discussion

Experiment 1

While there was virtually no evidence from field studies of predatory fishes feeding on non-indigenous fishes in the Ross River weirs (Chapter 4), Mouth almighty, at least under experimental conditions, readily consumed Mozambique mouthbrooder juveniles. Prey morphology, however, significantly affected prey selection by Mouth almighty as the predator showed a preference for Mozambique mouthbrooder much smaller than its gape width.

Field dietary studies showed that Mouth almighty fed on the most abundant indigenous fish species present in the Ross River weirs and dam. The absence of non-indigenous species (except for one Mosquitofish specimen), including Mozambique mouthbrooder juveniles, may reflect the low density of these fish in the habitat where Mouth almighty were sampled (margins of the main river channel). Unfortunately, insufficient samples of Mouth almighty were obtained for comparison of feeding preferences from locations where small non-indigenous fish were predominant (side channels and creeks blocked by dense vegetation).

While experimental and field data showed a positive correlation between predator size and prey size, there was a negative correlation between predator gape-width to prey body-depth (GW:BD) ratio: the relative size of prey in relation to the gape width of the predator decreased with increasing predator size. The electivity index data showed a clear preference by larger Mouth almighty for suitably sized Mozambique mouthbrooder prey that were much smaller than the theoretical maximum, the gape width of the predator. This supports observations that gape-limited predators do not necessarily select the largest prey available (Bence and Murdock 1986; Scott 1987; Schael *et al*. 1991; Paskowski and Tonn 1994).

These data suggest a learned response with age by predators to avoid risks of damage when handling spiny-finned prey. The effectiveness (strength) of Mozambique mouthbrooder fin spines increased significantly in fish with body depths between 10 and 15 mm size classes which corresponded with increasingly lower electivity index values (<+0.5) for Mouth almighty with gape widths above 15 mm. For Mouth almighty with gape widths less than 15 mm, the largest prey size-class with an electivity index greater than zero ranged between 89 and 93% of the predator gape width. These fish consumed Mozambique mouthbrooder juveniles which had weak or relatively few strong fin spines (particularly fish with body depths less than 10 mm). The smaller size class Mouth almighty consumed Mozambique mouthbrooder with body depths almost the same size as the predators' gape widths and were therefore able to maximise net energy return per prey with minimum handling costs of relatively large prey.

Above body depths of 15 mm, when all fin spines of Mozambique mouthbrooder were classed as "strong", the largest prey size class with an electivity index greater than zero ranged between 52 and 73% of the predator gape width. In this study, the experimental prey body depth as a percentage of predator gape width calculated for Mouth almighty was 65.2%, based on the mean prey body depths where the electivity index was >0.5. This value is within a range of 40-70% obtained from feeding studies for several predator species (see Werner 1974, 1977; Kislalioglu and Gibson 1976; Hoyle and Keast 1987; Chervinski *et al*. 1989), Wainwright and Barton (1995) concluded that ratios within this optimal range maximised the net energy return to the predator by minimising handling risks (e.g., fin spine injury) and energy costs (e.g., failure to swallow captured prey).

In the absence of environmental refuges, prey morphology (overall size, deep body and fin spines) can be an effective refuge from gape-limited predators for species such as Mozambique mouthbrooder. For small to medium sized predators, survivorship of adults of small species or larger-sized juveniles of large-bodied species with these attributes may therefore be enhanced (e.g., Blouw and Hagen 1984; Ajayi 1987; Bronmark and Miner 1992), while smaller-sized fish rely more on environmental refuges to avoid predation (Chapter 4 and Section *iv* below).

Experiment 2 and 3

Within the confines of the experimental mesocosms, in the absence of cover, prey capture time by predators may have been negligible in comparison with handling time, especially at higher prey densities. There was no difference in prey survivorship between ambush and patrol predators, but a significant difference in survivorship among prey species and between prey classes. In the time series experiment (Experiment 2), under the experimental conditions (low density, equal prey populations without prey replacement), predators, irrespective of foraging style, tended to first target slow and/or narrow-bodied fish in preference to fast, deep-bodied fish. Mosquitofish had the lowest survivorship and Agassiz's glassperch had the highest survivorship at all time intervals. Also, slow (non-indigenous) fish had significantly lower survivorship at 72 hrs compared with fast (indigenous) fish, and survivorship of narrow-bodied fish was significantly lower over the first 48 hrs compared with deep-bodied fish.

Prey morphology significantly affected survivorship with deep-bodied, spiny-finned fish (Mozambique mouthbrooder and Agassiz's glassperch), having consistently higher survivorship than fusiform fish with no or weak spines (Mosquitofish and Fly-specked hardyhead). Survivorship of Agassiz's glassperch was significantly higher than that of the Mozambique mouthbrooder: the species was more evasive and had stronger and larger fin spines for all size classes, while the Mozambique mouthbrooder was slower and only fish above 15 mm body depth had all fin spines classed as "strong". Only about 20% of Mozambique mouthbrooder used in the predation trials (10 mm body depth) had all strong spines. Also, for the two fusiform species, survivorship of Mosquitofish was significantly lower than that of the Fly-specked hardyhead, the latter being a faster-swimming, more evasive species compared with Mosquitofish.

In Experiment 3, for high density ("dominant") prey, survivorship of narrow-bodied, fusiform fish (Mosquitofish and Fly-specked hardyhead) was lower than that of deep-bodied, spiny-finned fish (Mozambique mouthbrooder and Agassiz's glassperch) for both ambush and patrol predators. However, there were similarly no significant differences in survivorship of prey species or prey classes between predator classes. In the experimental mesocosms, encounter (or capture time) for both types of predator may have been trivial compared with handling time especially with high-density prey. For low density ("non-dominant") prey, there was significantly lower survivorship of Mosquitofish with Patrol predators than ambush predators compared with the other three prey species, and there was significantly lower survivorship with Patrol predators of narrow-bodied fish (Mosquitofish and Fly-specked hardyhead) compared with deep-bodied fish (Mozambique mouthbrooder and Agassiz's glassperch). These differences, particularly the lower survivorship of "slow" (non-indigenous) species (Mosquitofish and Mozambique mouthbrooder) with patrol predators compared with ambush predators, may reflect differences in prey encounter rates between predator classes. These results are consistent with the model of predator-prey interactions of ambush and patrol predators by McCauley *et al.* (1993) that showed that the stability of a spatially structured predator-prey system depends on the relative mobility of prey and predators. In open space, with high prey and predator mobilities, greater amplitude in prey populations and even extinction of prey are common due to high encounter rates. They also found that decreasing predator and prey mobility introduces greater spatial heterogeneity: prey can, for example, "escape" into empty space and thus stabilise predator-prey systems.

These results also further suggest that these predators, irrespective of forage mode, are optimal foragers: that is, they reduce handling costs by feeding on narrow-bodied, soft-finned or slow prey in preference to deeper-bodied, spinous or fast, evasive prey. This preference is similar to observations made of experimental prey selection by other gape-limited predators such as Large- mouthed bass, *Micropterus salmoides* (Lewis *et al*. 1961; Hoyle and Keast 1987; Savino and Stein 1989; Hambright 1991), *Esox* spp (Wolfert and Miller 1978; Wahl and Stein 1988) and Eurasian Redfin perch, *Perca fluviatilis* (Christensen 1996). Gut content analyses of piscivores, including Barramundi and Tarpon, collected from the wild showed that these species frequently consume fusiform prey or deep-bodied prey with soft fins or weak fin spines (e.g., Chako 1956, Harrington and Harrington 1960; Russell and Garrett 1985; Coates 1987; Webb 1994). However, gape-limited predators, such as Barramundi and Tarpon, do eat deep-bodied and spiny-finned fish. Coates (1987) noted that spiny, deep-bodied species such as Glassperch were a minor component of the diet of Tarpon, while Davis (1985) and Russell and Garrett (1985) found that the species was an important component in the diets of smaller juvenile Barramundi from both fresh water and tidal reaches of creeks and from associated swamps in northern Australia. Danakusumah and Ismail (1986) reported Barramundi feeding in culture ponds on fusiform species but also on deep-bodied spiny forms such as *Tilapia (Oreochromis) mossambicus,* Glassperch (*Ambassis* spp) and Scats (*Scatophagus* sp.). Most of these studies, however, do not indicate whether predators are selecting particular size-class fish. Davis (1985), however, suggested that smaller ariid catfish were targeted as prey by Barramundi due to constraints imposed by the presence of large toxic fin spines. There was a distinct upper size limit for ariids with a maximum prey size (body length) of 35% of predator length, while, in contrast, the maximum length of Bony bream prey, a deep-bodied species without fin spines, was 61% of the predator length. Davis also commented that the presence of spines did not, on occasion, deter very large Barramundi as specimens were found with spines detached and embedded in the gut wall or even having penetrated the gut wall and found within the body cavity.

While there was no difference in prey survivorship between predator classes under the experimental conditions with high prey densities, under non-limiting conditions in open water, predator foraging mode may be more significant in terms of prey encounter and therefore prey selection. In open water, therefore, non-indigenous fish may be equally targeted as prey by predators as indigenous fish. Survivorship of non-indigenous (or indigenous) prey may, in part, depend on the type of predator present. However, irrespective of predator mode, prey survivorship in the presence of a predator is related to prey morphology (body depth and fin spines) or behaviour (evasiveness) rather than taxonomic identity.

Experiment 4

With access to cover in the presence of a predator, survivorship of all four prey species increased and was higher with increasing levels of cover. The most dramatic increase in survivorship was that of the two non-indigenous species, Mosquitofish and Mozambique mouthbrooder. At zero cover, Mosquitofish and Mozambique mouthbrooder had the lowest survivorship, but the highest survivorship at high cover. While survivorship of the two indigenous species increased with increasing level of cover, it was significantly lower than that of the non-indigenous species. The Fly-specked hardyhead had the lowest survivorship at high cover, while Agassiz's glassperch the second lowest. At zero cover, the Fly-specked hardyhead had the second highest survivorship and Agassiz's glassperch had the highest survivorship.

The two non-indigenous species were more effective than the indigenous species in their use of vegetation to avoid encounters with predators, although survivorship was higher for Agassiz's glassperch, which is a species more associated with vegetation than the pelagic Fly-specked hardyhead that prefers more open water. Kohde and Watanabe (1988) found that fish with horizontal stripes avoided vertically striped backgrounds, while those fish with vertical stripes would rest against vertically striped backgrounds. This colour morph (vertical barring) is characteristic of juveniles of many medium to large-sized cichlids, such as Mozambique mouthbrooder, that help to make them less visible among the vertical stems of plants (Baerends and Baerends-Van Roon 1950; Lanzing and Bower 1974). The horizontally-striped Fly-specked hardyhead tend to avoid vegetation and rely more on speed to avoid predation in open waters. In the experimental mesocosms without vegetation and against a grey background, both

Mosquitofish and juvenile Mozambique mouthbrooder rapidly became paler in colour, which would provide some camouflage, while among vegetation both species became darker probably also to further reduce predator detection in dense vegetation.

In the presence of cover, there were significant differences in survivorship of prey between predator classes. Ambush predators, notably the Mouth almighty, were more effective as predators among vegetation, which they use for concealment, compared with the patrol predators, such as Tarpon, that prefer to forage in more open water. Survivorship was highest for the two non-indigenous species, Mosquitofish and Mozambique mouthbrooder, particularly in high-density cover. Survivorship of both Agassiz's glassperch and the Fly-specked hardyhead also increased with increasing cover but was significantly lower than that of Mosquitofish and Mozambique mouthbrooder. These findings are consistent with other studies that demonstrated plant density affects predator-prey interactions: encounter rates with prey which enter vegetation as a refuge may decrease for patrol predators and increase for resident foragers (ambush predators), but generally decrease for all predators with increasing structural complexity (plant density) of the habitat (Crowder and Cooper 1979; Savino and Stein 1982, 1989a,b).

While dense vegetation may conceal prey from visual predators and be a rich source of food, high biological oxygen demand due to plant respiration at night can significantly lower dissolved oxygen to levels that can be lethal to both predatory fish and their prey (see Discussion section, Chapter 4). In the laboratory experiments, to minimise the risk of hypoxia to less tolerant fish, additional aeration was provided to the mesocosms in the small patches of open water in the high vegetation density trials. Fly-specked hardyhead aggregated in these patches of open water which probably made them more vulnerable to predation while the Mosquitofish, Mozambique mouthbrooder and Agassiz's glassperch dispersed through the vegetation. Unlike most indigenous species, Mosquitofish and Mozambique mouthbrooder, with access to surface water, are able to survive for extended periods in dense vegetation where hypoxic conditions can occur. Allen *et al.* (2002) noted that Glassperch may to some extent avoid hypoxic conditions as they tend to aggregate in and around vegetation during the day, but disperse to feed at night when extremely low dissolved oxygen levels are more likely to be experienced among vegetation and risk of detection by visual predators is lower. Where large concentrations of non-indigenous species such as poeciliids are present, competitive interactions may also make vegetation patches unfavourable habitat for indigenous species and make them more vulnerable to predation if they are forced into open water. The Mosquitofish, like other poeciliids, is an aggressive species and a fin nipper. This behaviour can result in mortality of attacked fish through secondary bacterial and fungal infection of the fins (Meffe 1983, 1985).

The species is also known to be a predator on the eggs and fry of other cyprinodontids and poeciliids overseas (Schoenherr 1981; Meffe 1985; Belk and Lydeard 1992) and of several Australian indigenous species (Ivantsoff 1999), most of which lay their eggs in or among vegetation.

Both Tarpon and Barramundi have been found to be effective predators in culture ponds of non-indigenous fishes where there is very little or no cover for prey species, but their effectiveness as predators is reduced where dense vegetation acts as either a physical or physiological (hypoxic) barrier. In the Ross River catchment, very few or no predators were found in the sites separated from the main Ross River weir channels by dense vegetation (see Chapter 4 Discussion). Non-indigenous fishes were the dominant fish component in samples collected from these sites, but were present, at least for the small-sized fish, in significantly lower numbers in samples from the main weir channels. In northern Queensland, the non-indigenous fish fauna consists of three established families: the Poeciliidae, Cichlids and Belontiidae and the family Cyprinidae, of which one unidentified cyprinid was reported, but of unknown status. All of these families have hypoxia-tolerant members. The poeciliids and cichlids are aquatic surface breathers and the belontiids are facultative air breathers with an auxiliary respiratory structure, the labyrinth organ, in the suprabranchial chamber. All of these species are therefore well adapted to hypoxic conditions created by the proliferation of aquatic vegetation.

Baltz and Moyle (1993) argued that predation has an especially important role in structuring stream fish assemblages and in resisting invasion by non-indigenous species as long as the streams are relatively undisturbed by human activity. The regional and local distribution patterns of non-indigenous species (Chapter 3 and 4) and the results of the survivorship trials in this study provide further support for these observations. Both ambush and patrol predators tested in the study are gape-limited, optimal foragers and generally preferred slow, fusiform, soft-finned species to deep-bodied, spiny-finned or evasive species as prey. Their choice of prey is therefore functional rather than taxonomic: in open water, they are as effective predators of non-indigenous species as they are of indigenous prey species. The Mosquitofish and the Mozambique mouthbrooder are two of the most widespread and abundant non-indigenous species in northern Queensland. The morphological characteristics of deep body and strong fin spines, particularly of larger juveniles and adults of large-bodied species, can provide an effective deterrent against predators, although smaller individuals are vulnerable to predation in the absence of environmental refuges. Ambush predators, such as the Barramundi and Mouth almighty, are more efficient predators among vegetation than patrol predators such as the Tarpon and Spangled perch. However, dense vegetation, characteristic of many highly modified agricultural and urban streams in northern Queensland, can act as a physical or physiological barrier to predatory fish irrespective of foraging mode. Such habitat can therefore provide an effective refuge for tolerant, non-indigenous fishes and increase the probability of their establishment and persistence.

This chapter has shown that:

- Adaptations of prey morphology (increased body depth and presence of large, strong spines) are effective against gape-limited predators
- under experimental conditions in open water, irrespective of prey density, there was no difference in survivorship of prey between predators with different foraging strategies;
- the predators investigated were optimal foragers, irrespective of foraging strategy and selected prey, that maximise net energy return by minimising capture and handling time; that is, they preferred slow, narrow-bodied and soft-finned fish (non-indigenous Mosquitofish) to fast, deep-bodied and spiny fish (Agassiz's glassperch); survivorship of fast-narrow-bodied fish (Fly-specked hardyhead) and slow, deep-bodied and spiny fish (Mozambique mouthbrooder) typically had intermediate survivorship between Mosquitofish and Agassiz's glassperch;
- ambush predators were more effective in capturing prey in vegetation than patrol predators, although capture efficiency of both types of predator significantly decreased in dense vegetation; and
- survivorship of the two non-indigenous fish species (Mosquitofish and Mozambique mouthbrooder) significantly increased with cover irrespective of density, with Mosquitofish, which had the lowest survivorship without cover, having the highest survivorship with cover, and Mozambique mouthbrooder the second highest survivorship with cover.

CHAPTER SIX

PARASITES OF NON-INDIGENOUS FISHES IN NORTHERN QUEENSLAND

> *"There are occasions when it is undoubtedly better to incur loss than to make gain"*
>
> Titus Marcus Plautus, ca. 184 BC
> from the play The Captives, Act II

6.1 Introduction

The introduction of an animal species into a new environment includes the introduction of a host-parasite system. Combes (1991) described this system in terms of two fundamental processes of parasite life cycles: compatability with one or more host species and likelihood of encountering particular host species. A "compatability filter" eliminates species that do not permit co-existence with the parasite for morpho-anatomical, metabolic or immunological reasons. An "encounter filter" excludes species that the parasite cannot meet for ecological or ethological reasons. Combes also noted that these filters are typically in flux if the relationship between host and parasite are modified. Ecological changes, such as entering novel habitat, will exert new selective pressures upon both the parasite and host to effect changes in, for example, parasite compatability, rates of encounter between parasite and potentially new hosts or host avoidance strategies.

Dogiel (1939) (in Petrushevski 1961) observed that "when an animal becomes acclimatised in an environment devoid of species to which it is closely related….its parasite fauna is strongly reduced" (p. 255). Other researchers (e.g., Petrushevski 1961; Dobson and May 1986; Bauer 1991, Kennedy 1994) subsequently expanded on Dogiel's original observations and Dove (1999) incorporated their results into a general model of host-parasite systems during the early phase of establishment in the receiving community (Figure 6.1). Petrushevski (1961) suggested that, during the early phase, the character and extent of changes in the host's parasite fauna often depend on the biotic and abiotic characteristics of the new environment. Dobson and May (1986) suggested that an introduced host species may find invading a new environment easier if the act of invasion reduces parasite fitness and increases the health of the host if, for example, the invader's usual parasites are left behind, or it does not encounter indigenous parasites, or it is not recognised by them as a suitable host. Dobson and May argued that the success of the introduced Starling and House sparrow in North America may be, partly, due to the smaller parasite fauna in their introduced range compared with their European homeland.

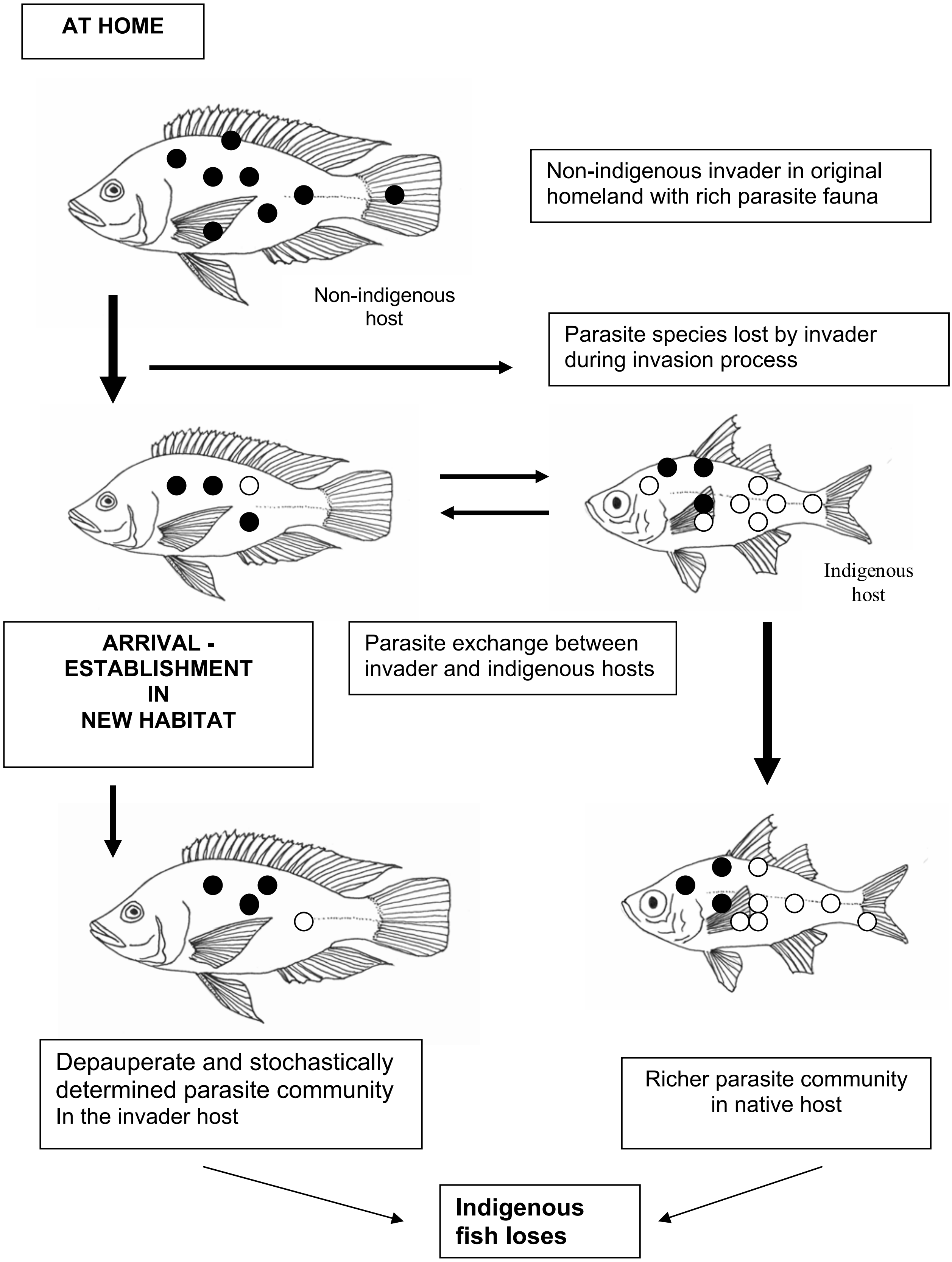


Figure 6.1 A generalised model of parasite richness processes during, and after, invasion by non-indigenous fishes (after Dove 1999)

They suggested that North American House sparrows might have benefited by the presence of relatively few blood parasites (common in European birds) which induce mortality or morbidity. North American Starlings might also have benefited by a reduction in helminth fauna as a result of eating fewer snails than their European counterparts (Dobson and May 1986).

Petrushevski (1961), Dobson and May (1986) and Bauer (1991) suggested that Direct Life Cycle (DLC) parasites are more likely to establish in a new environment (on invader or resident hosts) than Indirect Life Cycle (ILC) parasites that require intermediate hosts. The probability of establishment of ILC parasites is significantly increased if definitive and intermediate hosts are introduced together. Analysis of data from Hoffman (1970) by Dobson and May on the number of successfully invading species of DLC and ILC parasites of freshwater fish into Canada and Britain found that 75% were DLC parasites. Although the total number of unsuccessful DLC or ILC invaders was not known, Dobson and May inferred from the predominance of indigenous ILC parasites in both regions by a factor of 2 to 1, that the greater success of invading DLC parasites is significant. Bauer (1991) noted that introduced pathogens, including DLC and ILC parasites, could infect indigenous species that have little or no immunological defence, and cause significant mortality.

The probability of entry and establishment of a host may therefore be enhanced if potential predator or competitor populations are reduced by parasites or pathogens brought in with the invader. In novel encounters, the pathogenic potential of parasites may be influenced by the presence of taxonomically related hosts (in parallel transfer) or their absence (lateral transfer) in the new environment (Combes 1991; Bauer 1991). However, Combes (1991) noted that, while marked pathology and immune reactions are often a feature of recent encounters compared with “ancient” host-parasite systems, they do not necessarily occur in all host-parasite systems. The new host may not provide a suitable environment that maximises parasite fitness, including reproductive success and establishment of threshold levels of infection necessary for persistence of the parasite.

Maciolek (1984) described the negative impacts of introduced poeciliids and other exotic fishes as competitors and predators on the endemic fish fauna of the Hawaiian Islands. Font and Tate (1994) noted that exotic parasites, including *Bothriocephalus acheilognathi*, probably introduced with poeciliid hosts, represented a serious threat to indigenous fishes because of their known pathogenicity, either directly or as vectors of disease agents (Hoffman and Schubert 1984). Warner (1968) had suggested that extinction of about half the endemic land birds of Hawaii since 1778 was due to introduced diseases, including avian malaria and birdpox, following the introduction of a mosquito vector. According to Holmes and Price (1986), parasite community structure is a result of interactions among parasites and communities and can be located along an interactivity continuum, from interactive to isolationist. The former tend to be predictable and typically have high richness, large niche overlap, many core (high prevalence) species and a high level of similarity between infracommunities. The latter usually have the opposite characteristics and are largely stochastically determined. Recent studies (e.g., Kennedy and Bush 1994; Poulin 1997; Dove 1999) have examined changes in parasite community structure following introductions of fish hosts into new environments. These studies indicate that transfer of hosts results in a shift from more predictable, interactive parasite communities to more stochastically determined communities. Kennedy and Bush (1994), for example, found that for the salmonid, *Onchorrhyncus mykiss*, in its Canadian heartland, parasite communities were dominated by salmonid-specialist helminths forming a core phylogenetic (i.e., indigenous salmonid) element, with a minority ecological (i.e., other fish genera) element of broad generalists and non-salmonid specialists. Most parasite species except generic specialists were shared between host genera. As the distance to which *O. mykiss* was translocated from its heartland increased, generic specialists disappeared first and then salmonid specialists declined. The community was therefore increasingly composed of generalists and a lower total number of species. Poulin (1997) also reported a similar relationship between richness and parasite specificity for comparison of species rich and poor indigenous parasite faunas in Canadian fresh waters.

Guegan and Kennedy (1993) and Kennedy and Guegan (1994) noted that host residency times and spatial scaling need to be considered in assessing parasite communities of introduced fishes. For helminth parasites in Great Britain, they argued that, besides habitat and an omnivorous diet, parasite species richness is related to the time since the non-indigenous fish host's arrival. It is well established that there are latitudinal gradients in species diversity (Campbell 1993; Ricklefs and Miller 1999). Krebs (1978) stated that these gradients are a result of a complex interplay of factors over evolutionary time with tropical regions having greater productivity, competition and predation, and greater spatial heterogeneity and environmental stability with less extreme disturbance events than more temperate regions. According to Rohde (1991, 1992), there is greater effective evolutionary time (evolutionary speed) in the tropics that has resulted in greater species diversity through shorter generation times, faster mutation rates and faster selection at greater temperatures. Consequently, there are more "available niches" in the tropics than colder regions because of a greater number of free-living species which provide more opportunities for dependent species such as parasites due to greater effective evolutionary time (evolutionary speed). Rohde (1992) argued that differences in species richness between the same ecosystems at similar latitudes reflect differences in geological time. Rohde (1992) cited the example of monogenean species richness that is much greater in the cold-temperate waters of the older North Pacific Ocean compared with the younger North Atlantic Ocean. Kennedy (1995) found a greater parasite species richness and diversity for indigenous tropical Queensland anguillids compared with their temperate European counterparts. Kennedy suggested that the time hypothesis could provide a satisfactory explanation for the high diversity in the tropics of this ancient group.

Empirically, parasite dispersion patterns are often best described by the negative binomial distribution: some hosts have many parasites (over-dispersed) while most have just a few (Wilson *et al.* 1996; Shaw *et al.* 1998). Crofton (1971) considered the negative binomial distribution to be a fundamental model of parasitism and such aggregated distributions have been described for parasites on a wide range of hosts including invertebrates (Zhang *et al.* 1993; Westerman 1999), mammals (Gregory 1992; Kitamura *et al.* 1997; Magi *et al.* 2002), reptiles (Manweiler *et al.* 1992), birds (Rekasi *et al.* 1997; Kopocinski *et al.* 1998) and fish (Diamant 1989; Hallet and Roubal 1995; Nie and Yao 2000; Treasurer and Pope 2000). Crofton (1971) proposed that the negative binomial model for parasite dispersal within hosts can arise when: exposure to infection is random, but the probability of infection differs at each exposure (see also Nilssen and Haugerud 1995); infective stages are not randomly distributed (Keymer and Anderson 1979); infection increases the chance of further infection (e.g., changing habitat of host); the parasite life cycle is direct (see also Scott 1987; Poulin and Fitzgerald 1989; Brown *et al.* 2002); infection decreases the chances of further chances of infection (e.g., host immune response; density-dependent parasite or host mortality) (see Anderson and Gordon 1982; Hallett and Roubal 1995; Cognetti-Variale *et al.* 1996); variations in the host results in differences in the chance of infection (e.g., age, genotype and rate of development of host (see Halvorsen and Andersen 1984; Grenfell *et al.* 1995; Karlsbakk 2001); or there are temporal changes in chance of infection of host (see Kennedy 1987; Belghyti *et al.* 1994).

Numerous studies have demonstrated that parasites can have significant negative impacts on host fitness (i.e., the host's ability to survive and reproduce), for example, by direct mortality of the host (Chappell 1995), by host castration or inhibition of reproductive activity (Minchella *et al.* 1985; Dobson 1988; Guiyun, 1992), and by alteration of host behaviour so that the host is more susceptible to predation by predators that are definitive hosts of the parasite (Giles 1987; Poulin 1993; Lafferty 1999; Loot *et al* 2001). However, parasite aggregation can facilitate species coexistence in both parasite and host communities (Dobson and Roberts 1994; Roberts and Dobson 1995; Guiyun 1996; Morand *et al.* 1999; Pugliese 2000). Macinnis (1976) defined pathogenicity as a product of infectivity, parasite intensity and damage to the host, where infectivity was the combined host-finding capacity of the parasite and differential resistance of the host to invasion. Guiyun (1996) proposed that greater aggregation among hosts can reduce the ability of parasites to regulate host populations by limiting density-dependent pathogenic effects on hosts (due to more lightly infected than heavily infected hosts). Such aggregation can, in turn, influence the outcome of host competitive interactions. Guiyun (1996) argued that if individuals of a species are less susceptible to infection than the individuals of even a superior species and/or exhibit less pathology, then the otherwise inferior competitor will have a competitive advantage, which may result in stable coexistence or competitive exclusion of the previously superior competitor.

6.1.1 This study

This study investigated the parasite community structure of non-indigenous and indigenous freshwater fishes to assess the significance of any differences in the context of host fitness and post-invasion processes. Although host fitness was not directly measured in the study, it was hypothesised that hosts with lower individual parasite burdens, fewer infected individuals and more aggregated parasite distributions among hosts will be advantaged by lower energetic costs of parasitism and risk of mortality and therefore increased probability of surviving to reproduce (i.e., increased fitness).

Dove (1999) found that introduced poeciliids in southeastern Queensland had depauperate parasite communities and argued that this conferred on these fishes a competitive advantage in relation to indigenous fishes. Besides parasite species richness, parasite aggregation patterns can significantly influence community structure, although no studies have examined such patterns in non-indigenous hosts during the initial phase of establishment in a new habitat. This study presented an opportunity to investigate these patterns and to consider how they might influence the successful establishment of host-parasite systems in a new habitat.

No previous studies have examined residence time of non-indigenous fishes in relation to the structure of their parasite communities, nor considered its importance in relation to host fitness. It is hypothesised that a slow accumulation of parasites may advantage non-indigenous fishes with depauperate parasite communities by maintaining lower parasite burdens over a long time period and further reduce the probability of adverse effects of parasites on host fitness. There are no estimates of parasite acquisition by non-indigenous fishes either in tropical or temperate regions. Questions addressed by this study, therefore, were: is the rate of parasite acquisition by non-indigenous fishes in new habitat a relatively slow process; and are there latitudinal differences in parasite acquisition – do non-indigenous fishes in the tropics acquire parasites at a faster or slower rate than their temperate counterparts?

A number of studies have demonstrated that environmental stressors can alter the host-parasite interaction through changes either in host susceptibility to infection (through compromised immune response) (Oldewage and van As 1987; Radheyshyam *et al.* 1993; Mellergaard and Nielsen 1995; Hoole 1997; Iida and Kurogi 2001; Jeney *et al.* 2002) and/or by changes in parasite fauna (affected by changes in water quality, or by loss or proliferation of intermediate hosts) (Marcogliese and Cone 1996, 1997; D'Amelio and Gerasi 1997; Lafferty 1997; Landsberg *et al.* 1998). Dove (1999) demonstrated that habitat disturbance could significantly reduce parasite species richness on non-indigenous, poeciliid hosts. Most non-indigenous fishes in Australia occupy habitats suffering from disturbance, particularly in urban and agricultural environments, often close to the sites of initial introduction (Chapter 3). Declines in indigenous fish communities and changes to habitat (Chapter 4) have benefited non-indigenous fish by reduction in resource competition and predation pressure but may also benefit by changes in survival or transmission of parasites due to anthropogenic alteration of aquatic habitats.

The aims of this study, therefore, were to evaluate the parasite fauna of non-indigenous fishes in northern Queensland in relation to current models and concepts of parasitism in introduced species, and to consider whether any changes in parasite community structure related to post-invasion processes enhance the fitness of non-indigenous fishes and their ability to compete with indigenous fishes and to establish self-maintaining populations. Specifically the objectives were to:

- describe the parasite fauna of non-indigenous fishes in northern Queensland;
- compare the parasite community structure of non-indigenous and indigenous host fish species and assess the degree of interactivity of these communities;
- describe the dispersion patterns of parasites in non-indigenous and indigenous host fish species in northern Queensland;
- estimate and compare rates of parasite acquisition in relation to residence time by non-indigenous host fish species in tropical northern Queensland with data for non-indigenous fishes from temperate regions of southern Australia and Great Britain.
- compare parasite richness, intensity and prevalence and dispersion patterns between non-indigenous and non-indigenous fish species in relation to levels of habitat disturbance; and
- evaluate how these data fit with existing models of parasite community structure and acclimatisation of host-parasite systems in new environments.

6.2 Methods

6.2.1 Data collection

Sampling locations and methods

Non-indigenous fish samples were collected from coastal and inland freshwater sites in northern Queensland between the Burdekin and Daintree River catchments (see Chapter 3 and Table B1, Appendix B). Samples of indigenous freshwater fish were collected mainly from coastal freshwater sites located in the Townsville region north to Ingham and south to Giru. Samples of non-indigenous and indigenous fish collected in gill nets from the Ross River weirs (see Chapter 4) were also examined for parasites.

For small-bodied fish collected in seine and dip-nets, samples were transported live in large aerated and insulated containers to the laboratory as soon after capture as possible. Fish were euthenased by placing them individually in a water-filled plastic bag, which was then suspended in an ice slurry. Previous testing indicated that this treatment and examination procedure did not result in immediate loss of ectoparasites from the host.

Fish were collected at dusk and, after removal from the nets, briefly examined to determine the location of any large ectoparasites. Fish were placed separately in large plastic bags containing water and then euthenased by placing the bag in an ice slurry. Fish samples were then transported to the laboratory and examined immediately or kept in a cold store overnight and examined the following day. The contents of the plastic bags were also carefully examined.

Host examination and parasite identification

Fish were examined externally and internally for macroparasites (i.e., parasites large enough to be detected directly or under low power magnification). These included large protists, such as whitespot, helminths (monogeneans, digeneans and nematodes) and crustaceans (copepods and branchiurans). Microparasites, such as blood or other intracellular parasites, and very small protists, such as trichodinids, were not included in the study. The total number of each species of parasite, their location on the host, the total length of the host and number of hosts examined were recorded.

The external body surface and fins were examined under a stereo dissector microscope and ectoparasites were removed by microprobes and pipettes. Gills of each fish were removed and placed in freshwater and examined under a dissector microscope. Eyes, brain, body wall musculature and internal organs, including the gut and intestines, were each examined separately.

Monogeneans collected from gill filaments and external body surfaces were transferred live in water to microscope slides and allowed to dehydrate then fixed in Malmberg's solution (ammonium picrate glycerin) (Malmberg 1970). Slide covers were sealed with clear nail polish and parasites were examined at high power using a compound microscope.

Helminths were removed from the host and fixed in calcium-acetate-buffered formalin (70%) and stored in 70% ethanol. Digeneans and cestodes were stained with dilute acetocarmine and then dehydrated through a graded ethanol series, cleared in xylene and mounted in Canada balsam. Nematodes and acanthocephalan specimens were examined as wet preparations after clearing in lactophenol mountant.

Parasites were identified to the lowest taxonomic level possible using current taxonomic literature (see Appendix E, Table E1) and with invaluable assistance from the following parasitologists in Australia and overseas: Dr W. Poly (Department of Ichthyology, California Academy of Sciences, San Fransisco, USA), Dr A. Dove (Marine Science Research Centre, Stony Brook University, NY, USA), Dr Diane Barton (Zoology Department, JCUNQ, Townsville, Australia), D. Corlis (Zoology Department, JCUNQ, Townsville, Australia) and A. Fletcher (Cairns Port Authority, Cairns, Australia).

Designation of parasites as either indigenous or non-indigenous species was based on current taxonomic and ecological literature and data collected during the study, including the known distribution of hosts in Australia and overseas, occurrence of larval stages of intermediate life cycle (ILC) parasites and the prevalence and intensity of each species, especially where identification was to generic level or higher. For example, where a parasite occurred in very low prevalence and intensity in a non-indigenous host and significantly higher in an indigenous host, it was more probable that the parasite was an indigenous species and that the non-indigenous infections were stochastic events.

6.2.2 Data analysis

i Summary statistics, regional parasite frequency distributions on host and parasite species richness

Summary statistics were obtained for indigenous and non-indigenous fish parasite communities. Data were tested for normality and then analysed with the appropriate parametric or non-parametric tests.

Regional parasite species frequency distributions were generated and parasite species richness was compared between indigenous and non-indigenous hosts using the Mann-Whitney U test. The goodness of fit of the species frequency distributions to the Negative Binomial model was done using a χ^2 test (see procedure below). Association between host size and parasite intensity for non-indigenous fish and indigenous fish was examined by calculating Pearson correlation coefficients. Parasite intensity and prevalence for non-indigenous and indigenous fish were compared using the Mann-Whitney U test.

A χ^2 goodness of fit to the Negative Binomial model for parasite species richness and parasite intensity data was done using the following formula from Fowler *et al.* (1998) to calculate expected frequencies:

Negative binomial distribution:

$$P_{x=0} = (1 + x/k)^{-k}$$

$$P_{x=1} = k \times F \times P_{x=0}$$

$$P_{x=n} = \frac{(k+[n-1]}{n} \times F \times P_{x=n-1}$$

where $k = x/(s^2 - x)$ and $F = x/(x + k)$, and x = sample mean and s^2 = sample variance.

The k parameter has been widely used as a parasite aggregation index (e.g., Poulin 1993; Shaw and Dobson 1995; Shaw *et al.* 1998), including its use with fish hosts (e.g., Xiao-qin *et al.* 1999; Ting-bao *et al.* 2000; Nie and Yao 2000; Treasurer and Pope 2000). As k increases (and the variance decreases relative to the mean), the distribution becomes less positively skewed and more symmetrical. The distribution, therefore, is less aggregated when the prevalence of infection and intensity (mean number of parasites per host) increase. Since values of k can also be associated with different means and *vice versa* (Pennyciuck 1971), direct comparisons of k need to be considered in terms of the mean intensity and prevalence of the respective host-parasite systems (Scott 1987; Grafen and Woodhouse 1993).

Data were tested for normality and appropriate parametric or non-parametric tests applied. The non-parametric Mann-Whitney U test was used to compare indigenous and non-indigenous host lengths, parasite aggregation (k), parasite richness and intensity between habitat disturbance levels. Pearson correlation (r) coefficients were calculated to examine the relationship between host length and intensity for indigenous and non-indigenous fish for each habitat disturbance level. Parasite frequency distributions for indigenous and non-indigenous fish and for individual fish species in each habitat type were also tested for goodness of fit to the negative binomial model as described above.

ii. Parasite fauna similarity/difference analyses

Core-satellite species

Regional parasite community structure for non-indigenous and indigenous host species was also defined in terms of the core-satellite species concept (Hanski 1982; Gibson *et al.* 1999) where species fall into three main categories: satellite, intermediate and core species. Satellite species are rare with low prevalence, while core species are regionally common with high prevalence. From prevalence data, the numbers of ‘core’, ‘intermediate’ and satellite’ parasite species for indigenous and non-indigenous hosts were determined.

The different categories of parasite species were defined in terms of prevalence as follows:

- Core species : $\geq 70\%$
- Intermediate species : between 20 and 70%
- Satellite species : $\leq 20\%$

Non-metric multi dimensional scaling (NMDS) ordination analysis

NMDS ordination analysis using PCORD (McCune and Metford 1999) was performed with a presence/absence input matrix for parasites on hosts (Table E2, Appendix E) to examine the degree of similarity or difference between parasite assemblages of indigenous and non-indigenous host fishes.

Cumulative parasite species curves and parasite community interactivity

Interactivity is the degree to which parasite species interact with each other to create structure in a parasite community (Dove 1999). It is usually described as a continuum from interactive to isolationist communities (Holmes and Price 1986). Interactive communities typically have high species richness, large niche overlap, many core parasite species and a high level of similarity between infracommunities (parasite populations within a single host). Isolationist communities typically have the opposite characteristics and are, therefore, largely stochastically-determined. Assessment of interactivity can be obtained from cumulative parasite species curves and calculation of interactivity indices using a similar procedure described by Dove (1999). The cumulative species curves were fitted with 2-parameter Weibull growth curves of the form:

$$y = a - ae^{(-bx)}$$

where a = curve asymptote (predicted maximum component community richness), b = rate at which the asymptote is reached (index of mean infracommunity richness that reflects the number of species per host and prevalence), y = cumulative regional richness and x is the cumulative sample size. Curves were fitted using non-linear regression in Sigma Plot and r and r^2 values were calculated for goodness of fit for each curve. Curves were plotted using a cumulative percentage of the total parasite fauna for each species on the y-axis to represent the rate at which parasite "saturation" of the host species was reached.

An interactivity index G was calculated using the above values of *a* and *b,* where G = *a.b* (Dove 1999), and *a* typically has a value in tens of species, while *b* has a value between 0 and 1. Their product usually gives a value between 0 and 5. According to Dove (1999), for individual host species, more interactive parasite communities tend to have higher G values. In intermediate communities between highly interactive (high *a, b*) and isolationist (low *a, b*), the b parameter (index of mean infracommunity richness) tends to be more important in determining interactivity. A high degree of similarity of infracommunities will result in more saturated, predictable and interactive component communities (all of the infracommunities within given host populations).

Parasite acquisition by non-indigenous hosts in relation to residence time

Linear regression analyses were undertaken using SPSS to examine the relationship between parasite acquisition by non-indigenous hosts and their residence time in northern Queensland and also, for comparative purposes, for non-indigenous fishes in temperate regions of southern Australia, and Great Britain (Figure E1, Appendix E). Residence times were based upon known or estimated time of introductions of non-indigenous fish species from this study, personal records and from the literature (Maitland 1972; Wheeler 1977; McKay 1978; Kennedy 1993; McDowall 1996; Phillips and Rix 1985; McKay 1984; Lear 1987; Webb 1994; Anon. 2003a,b). Parasite data were obtained from this study, from the literature (Kennedy 1974; Backhouse and Gooley 1979; Beumer *et al.* 1982; Langdon 1987; Guegan and Kennedy 1993; Webb 1994; Lom and Dykova 1995; Willoughby 1998; Gaze and Wootten 1998; Dove 1999; D. Morgan, Centre for Fisheries research, Murdoch University, W.A. *pers. comm*. The regression slope (a) provides an index of parasite acquisition rate and r^2 coefficient the strength of the linear relationship. The predicted residence time for a generalised host to acquire a regional parasite community of a given number can be obtained from the regression equation:

N(number of parasites acquired) = Constant (y intercept) + a (slope) × Residence time

Residence estimates were made for a “generalised” invader to acquire a parasite fauna of 15-25 species, similar to the range obtained for common indigenous fish species (Empire gudgeon, Agassiz’s glassperch, Eastern Queensland rainbowfish and Fly-specked hardyhead) in northern Queensland. Estimates were made using data obtained for tropical and temperate hosts in Australia and Britain for comparison. Residence estimates for the Brown trout using temperate Australian and Great Britain regression data were obtained for comparison with the known residence time of Brown trout in Britain (10,000 yrs B.P. – Wheeler 1977) based on a parasite fauna of at least 50 species acquired by this host in Britain (Kennedy 1974). The comparison of an estimate of Brown trout residency time extrapolated from these data with known residency time would indicate if the relationship between residency and acquisition is linear over much longer time scales, i.e., from decades to millenia).

Comparison of parasite faunas of non-indigenous and indigenous hosts between high disturbance and low disturbance habitats

Parasite frequency distributions and population parameters (species richness, intensity, prevalence, aggregation (k) parameter) were compared using the non-parametric Mann-Whitney U test between indigenous and non-indigenous hosts collected in "high disturbance" and "low disturbance" habitats (see Table B1, Appendix B for sites where host fish samples were collected).

High disturbance habitats were distinguished by absence of riparian vegetation, extensive invasion or overgrowth by terrestrial or aquatic weeds, and obvious instream pollution (e.g., oil). Examples of low disturbance habitats were Canal Creek, Thuringowa (Figure 6.1) and Majors Creek, Woodstock (Figure 6.2) with largely intact riparian vegetation and little instream vegetation. Examples of high disturbance habitats were: Cranbrook Creek, Townsville, an urban stream modified for stormwater drainage with little riparian vegetation intact and extensive overgrowth of aquatic vegetation (Figure 6.3), and natural flood channels modified into water hazards on the Rowes Bay Golf Course, Pallarenda, with no riparian vegetation, modified banks and channel beds and organically polluted with large, scattered patches of submerged aquatic vegetation present (Figure 6.4).

Figure 6.2 Canal Creek, Thuringowa, an example of a Low disturbance habitat

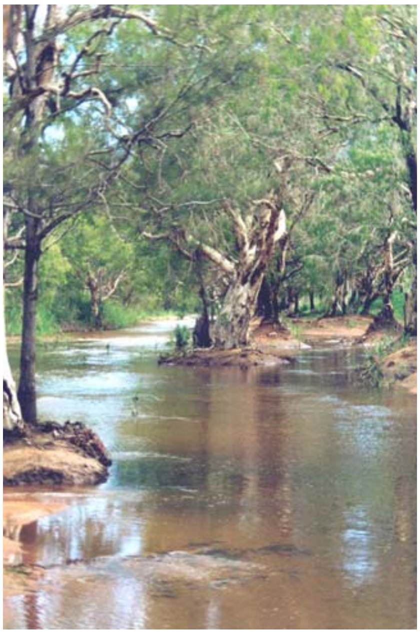


Figure 6.3 Majors Creek, Woodstock, an example of a Low disturbance habitat

Figure 6.4 Cranbrook Creek, Cranbrook, Townsville. The creek has been modified to form a stormwater drain (note: outlet pipes in mid-background), an example of a High Disturbance habitat

Figure 6.5 Rowes Bay Golf course, natural flood channels modified to form water hazards, an example of a High disturbance habitat

6.3 Results

6.3.1 Parasite fauna

A total of 3269 parasites were recorded from 1928 non-indigenous host fish of 11 species. These consisted of six small-bodied species (Burton's haplochromis, Jewel cichlid, Mosquitofish, Guppy, Swordtail, Platy, Three-spot gourami) and juveniles and adults of the large-bodied Mozambique mouthbrooder, as well as adults of other large-bodied cichlids (Black-spotted mangrove cichlid, Oscar and Midas cichlid). Parasites were found on all species except the Three-spot gourami (Table 6.1) A total of 9905 parasites were recorded from 830 indigenous host fish of six species. There were five small-bodied species (Agassiz's glassperch, Eastern Queensland rainbowfish, Fly-specked hardyhead, Purple-spotted gudgeon and Empire gudgeon), and juveniles of one medium-sized species, the Mouth almighty (Table 6.2). These fish were collected from sites in coastal northern Queensland between the Burdekin and Daintree River catchments and from the Atherton Tablelands, with the largest number of specimens collected in the Townsville-Thuringowa region (see Table B1, Appendix B for sampling site locations).

For non-indigenous fish, a total of 22 parasite species were identified (Table 6.1), with 16 taxa designated as indigenous species and five as non-indigenous species, and with one protist species, the Whitespot, *Ichthyophthirius multifiliis*, classed as cosmopolitan. Excluding Whitespot, nine species were direct life cycle (DLC) parasites with simple life cycles and 12 species were intermediate life cycle (ILC) parasites with complex life cycles. The majority of the ILC parasites were intermediate, larval stages with low host specificity, while all of the DLC parasites included adults with host specificity ranging from low to high depending on origin (indigenous or non-indigenous). The indigenous DLC parasites found on non-indigenous hosts, e.g., *Argulus* sp. 1, had low host specificity, while the non-indigenous DLC parasites, e.g., monogeneans, had high host specificity. The ratio of DLC to ILC parasite species was 0.75, with DLC parasites only 42.9% of the total parasites recorded (Table 6.3).

For indigenous fish, a total of 32 parasite species were identified (Table 6.2). All parasites were designated as indigenous species. The ratio of DLC to ILC species was 0.88 with DLC parasites only 47% of the total parasites recorded (Table 6.3). For the non-indigenous hosts, four of the five non-indigenous parasite species were monogeneans, which were most probably introduced with their respective hosts. This is the first report in Australia of the monogenean, *Cichlidogyrus tilapiae,* on Mozambique mouthbrooder. The monogenean, *Urocleidoides* sp. 1., found on the Platy is very similar to specimens tentatively assigned to "cf. *Leptocleidus* sp ." by

Table 6.1 Parasites collected from non-indigenous fishes in northern Queensland fresh waters (n = indigenous; e = non-indigenous; Spec. = host specificity; ecto = ectoparasite; endo = endoparasite)

Host *	Parasite species	Taxon	n/e	Spec.	Location on host	
PR	*Gyrodactylus bullatarudis*	*Monogenea*	ex	high	ecto	fins, body
	Telosentis sp. 1	Acanthocephala	n	low	endo	intestine
	Metacercarial cyst sp. 3	Digenea	n*	low	endo	body cavity
	Dracunculid (Spirurida) sp. 1	Nematoda	n	low	endo	liver
	Contracecum sp. 1	Nematoda	n	low	endo	intestine
	Bothriocephalus acheilognathi	Cestoda	ex	low	endo	intestine
	Clinostomum sp.(*complanatum*)	Digenea	n	low	endo	body cavity
	Amphilinidea sp.1	Cestoda	n*	?	endo	body cavity
	Hydracarina sp. 1	Arachnida	n	low	ecto	body
GAM	*Salsuginus heterocliti*	Monogenea	ex	high	ecto	gills
	Echinostome sp. 1	Digenea	n	low	ecto	gills
	Clinostomum sp.(*complanatum*)	Digenea	n	low	endo	body cavity
	Metacercarial cyst sp. 2	Digenea	n*	low	endo	liver
	Metacercarial cyst sp. 3	Digenea	n*	low	endo	body cavity
	Dracunculid (spirurida) sp. 1	Nematoda	n	low	endo	liv., b. cavity
	Contracecum sp. 1	Nematoda	n	low	endo	intestine
	Telosentis sp. 1	Acanthocephala	n	low	endo	intestine
	Glochidium sp. 1	Mollusca	n	low	ecto	gills
XM	*Urocleidoides* sp. 1	Monogenea	ex	high	ecto	gills
	Ichthyophthirius multifiliis	Protista	n	low	ecto	body, fins
	Philometriodes sp. 1	Nematoda	n	low	endo	operculum
	Hydracarina sp. 1	Arachnida	n	low	ecto	body
XH	*Ichthyophthirius multifiliis*	Protista	n	low	ecto	body, fins
MM	*Cichlidogyrus tilapiae*	Monogenea	ex	high	ecto	gills
	Telosentis sp. 1	Acanthocephala	n	low	endo	intestine
	Argulus sp. 1	Branchiura	?	low	ecto	body, fins
	Ichthyophthirius multifiliis	Protista	n	low	ecto	body, fins
	Clinostomum sp.(complanatum)	Digenea	n	low	endo	body cavity
	Metacercarial cyst sp. 2	Digenea	n*	low	endo	body cavity
	Metacercarial cyst sp. 3	Digenea	n*	low	endo	liv., b. cavity
	Echinostome sp. 1	Digenea	n	low	ecto	gills
	Bothriocephalus acheilognathi	Cestoda	ex	low	endo	intestine
	Eustrongylid sp. 2	Nematoda	n	low	endo	operculum
	Dracunculid (Spirurida) sp. 1	Nematoda	n	low	endo	liver, b. cavity
TM	*Ergasilus australiensis*	Copepoda	n	low	ecto	gills
AC	*Argulus* sp. 1	Branchiura	?	low	ecto	body, fins
	Eustrongyloides cf excicus	Nematoda	n	low	endo	body cavity
AO	*Argulus* sp. 1	Branchiura	?	low	ecto	body, fins
HB	Echinostome sp. 1	Digenea	n	low	ecto	gills
HG	*Ichthyophthirius multifiliis*	Protista	n	low	ecto	fins

Hosts: PR = Guppy; GAM = Mosquitofish; XM = Platy; XH = Swordtail; MM = Mozambique mouthbrooder; TM = Black-spotted mangrove cichlid; AC= Midas cichlid; AO = Oscar; HB = Burton’s haplochromis; HG = Jewel cichlid

Table 6.2 Parasites recorded from indigenous fishes in northern Queensland fresh waters (n = indigenous; e = non-indigenous; Spec. = host specificity; ecto = ectoparasite; endo = endoparasite)

Host	Parasite species	Taxon	n/e	Spec.	Location on host	
RF	Longidigitis iliocirrus	Monogenea	n	high	ecto	gills
	L. auripontiformis	Monogenea	n	high	ecto	gills
	L. chunkyanchor	Monogenea	n	high	ecto	gills
	Helicirrus splendida	Monogenea	n	high	ecto	gills
	H. megaanchor	Monogenea	n	high	ecto	gills
	Clinostomum sp.(*complanatum*)	Digenea	n	low	endo	body cavity
	Metacercarial cyst sp. 1	Digenea	n	low	endo	liver
	Metacercarial cyst sp. 2	Digenea	n	low	endo	body cavity
	Echinostome cyst sp. 1	Digenea	n	low	ecto	gills
	Taenid sp. 1	Cestoda	n	?	endo	body cavity
	Telosentis sp. 1	Acanthocephala	n	low	endo	intestine
	Myxozoan sp. 2 (cf. Hennegrya)	Protista	n	?	ecto	gills
	Dracunculid (Spirurida) sp. 1	Nematoda	n	low	endo	liver, body cavity
	Philometrid sp. 1	Nematoda	n	low	endo	body cavity
	Contracecum sp. 1	Nematoda	n	low	endo	body cavity
HH	*Longidigitis curvatus*	Monogenea	n	high	ecto	gills
	Recurvatus chelatus	Monogenea	n	high	ecto	gills
	Ecinostome sp. 1	Digenea	n	low	ecto	gills
	Metacercarial cyst sp. 1	Digenea	n	low	endo	liver
	Metacercarial cyst sp. 2	Digenea	n	low	endo	liver
	Contracecum sp. 1	Nematoda	n	low	endo	body cavity
	Philometrid sp. 1	Nematoda	n	low	endo	operculum
	Taenid sp. 1	Cestoda	n	low	endo	liver
GP	*Haplocleidus* sp. 1	Monogenea	n	high	ecto	gills
	Echinostome sp. 1	Digenea	n	low	ecto	buccal cavity
	Clinostomum sp.(*complanatum*)	Digenea	n	low	ecto	body cavity
	Metacercarial cyst sp. 1	Digenea	n	low	endo	liver, caudal fin
	Metacercarial cyst sp. 2	Digenea	n	low	endo	body cavity
	Metacercarial cyst sp. 3	Digenea	n	?low	endo	body cavity
	Metacercarial cyst sp. 4	Digenea	n	?low	endo	liver
	Eustrongyloides sp. cf. *excisus*	Nematoda	n	?	endo	body cavity
	Glochidium sp. 2	Mollusca	n	?low	ecto	fins
EG	*Pseudodactylogyroides kaligaensis*	Monogenea	n	h	ecto	gills
	Clinostomum sp.(*complanatum*)	Digenea	n	low	endo	body cavity
	Echinostome sp. 1	Digenea	n	low	ecto	gills
	Metacercarial cyst sp. 1	Digenea	n	low	endo	liver
	Metacercarial cyst sp. 3	Digenea	n	low	endo	body cavity
	Metacercarial cyst sp. 4	Digenea	n	low	endo	body cavity
	Spirocamallanus sp. 1	Nematoda	n	?	endo	intestine
	Contracecum sp. 1	Nematoda	n	low	endo	body cavity
	Taenid sp. 1	Cestoda	n	?	endo	liver
	Trichodinid sp. 1	Protista	n	high	ecto	gills
PSG	*Pseudodactylogyroides kalingaensis*	Monogenea	n	high	ecto	gills
	Contracecum sp. 2	Nematoda	n	?	endo	body cavity
	Clinostomum sp.(*complanatum*)	Digenea	n	low	endo	body cavity
	Metacercarial cyst sp. 5	Digenea	n	?	endo	body cavity
	Taenid sp. 1	Cestoda	n	?	endo	intestine
MA	*Pseudodactylogyroides khloensis*	Monogenea	n	high	ecto	gills
	Ergasilus intermedius	Copepoda	n	low	ecto	gills
	Echinostome sp. 1	Digenea	n	low	ecto	gills
	Metacercarial cyst sp. 1	Digenea	n	low	endo	liver
	Metacercarial cyst sp. 2	Digenea	n	low	endo	liver
	Clinostomum sp.(*complanatum*)	Digenea	n	low	endo	body cavity
	Taenid sp. 2	Cestoda	n	?	endo	liver
	Philometrid sp. 1	Nematoda	n	?low	endo	int. operculum
	Spirocamellanus sp. 1	Nematoda	n	?	endo	intestine
	Spinitectus sp. 1	Nematoda	n	?	endo	foregut, intestine
	Contracecum sp. 1	Nematoda	n	low	endo	body cavity

Hosts*: RF = Eastern Queensland rainbowfish; HH = Fly-specked hardyhead; GP = Agassiz's glassperch; EG = Empire gudgeon; PSG = Purple-spotted gudgeon; MA = Mouth almighty

Dove (1999). Dove first reported the species on the Platy and Swordtail from southern Queensland waters.

There is no information on monogenean parasites on the Platy in its indigenous range and this species' original host may be another poeciliid. Dove first reported the monogenean, *Gyrodactylus bullatarudis*, on the Guppy collected from the Barron River, in northern Queensland, and from southern Queensland waters, and also the monogenean*, Salsuginus seculus,* on Mosquitofish from southern Queensland waters. No non-indigenous parasites were recorded from indigenous fish hosts in the study.

The cichlids and poeciliids shared five indigenous parasites: the acanthocephalan, *Telosentis* sp. 1, four digenean metacercaria (encysted sp. 2 and 3, echinostome sp. 1 and *Clinostomum complanatum*), and one cosmopolitan parasite, the protist, *Ichthyophthirius multifiliis* (White-spot).

This is the first report on cichlid fishes in Australian fresh waters of: the acanthocephalan, *Telosentis* sp. 1; the nematode spirurid sp. 1; *Contracecum* sp. 1; the two cestode species (Amphilinid sp. 1 and *Bothriocephalus acheilognathi*); the digenean metacercaria (sp. 3 and *Clinostomum complanatum*); and the protist, *Ichthyophthirius multifiliis*. The acanthocephalan, Telosentis sp. 1, is a new species (D. Barton, *pers. comm.)* and the indigenous host appears to be the Eastern Queensland rainbowfish, although the parasite's host specificity is not known. The branchiuran argulid ectoparasite, found on adult Mozambique mouthbrooder, the Oscar and Midas cichlid, is a new species (W. Poly, *pers. comm.*) with new host species records. The species appears to have low host specificity as it was also found on large-bodied indigenous fish species, including the Barramundi, Freshwater longtom, Sleepy cod and Black catfish, collected by gill nets in the Ross River weirs. This is also the first record of an ectoparasitic copepod, *Ergasilus australiensis*, on a non-indigenous fish in Queensland fresh waters, and the first report of the species on the Black-spotted mangrove cichlid (*Tilapia mariae*). Excluding *Gyrodactylus bullatarudis*, this is the first report of the other monogenean species from poeciliid fishes in northern Queensland waters. This is also the first report of an acanthocephalan from introduced poeciliids in Australian waters.

This is the first report of the non-indigenous cestode, *Bothriocephalus acheilognathi*, in northern Queensland waters and the first report of this parasite from the Mozambique mouthbrooder and the Guppy in Australian waters. The parasite was found in fish from only one site (Wright Creek, south of Cairns). The principal host of this parasite is the European carp, although there have been no reports of this species in open waters in northern Queensland (except possibly for the unidentified cyprinid sp. A from the Ross River in 2003 - Chapter 3). The immediate origins of the parasite are not known, although it may have been introduced with infected fish kept locally in ponds. Infection of indigenous fish by this parasite was not determined as none were sampled from this creek.

6.3.2 Parasite community structure

Species richness

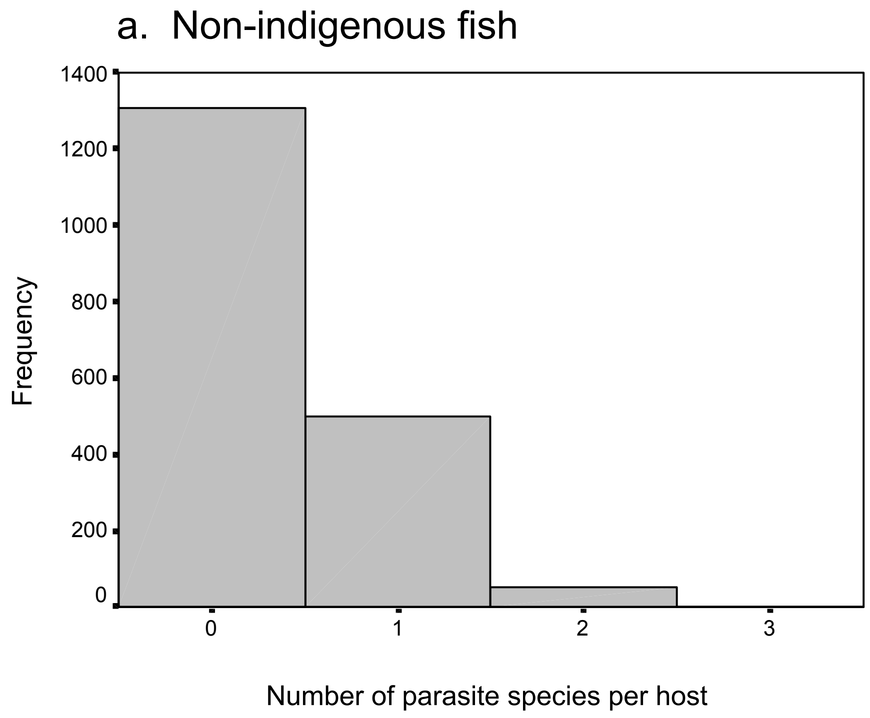
Summary statistics for parasite faunas of indigenous and non-indigenous fishes are presented in Table 6.3. The median parasite richness for non-indigenous fishes (0) was significantly lower than that for indigenous fishes (2) (U = 193625.5; $p < 0.001$). The parasite richness frequency distribution curve was more negatively skewed for non-indigenous fishes (Figure 6.6a). The curve for non-indigenous fishes fitted the negative binomial distribution ($\chi^2 = 20.111$; df = 16; p = 0.215), while that for indigenous fishes differed significantly from it ($\chi^2 = 165.834$; df = 43; $p < 0.001$) (Figure 6.6b). Total parasite species prevalences for non-indigenous species were much lower compared with indigenous species with proportionately greater numbers of non-indigenous fishes with no parasites found compared with indigenous fishes. Mean parasite species richness for indigenous fishes was 1.8, with a maximum of seven parasite species recorded from the Eastern Queensland rainbowfish, while the mean richness for non-indigenous fishes was 0.3, with a maximum of three parasite species recorded from the Guppy. Total prevalence for non-indigenous fishes was 28.4% while the prevalence for indigenous fishes was 90.0%.

Median host size was significantly larger for indigenous fishes (3.6 cm) compared with non-indigenous fishes (3.1 cm) (U = 391044.5; $p <.001$) although the difference was relatively small (0.5 cm) and overall size ranges for the two groups of fishes were very similar. The size range for indigenous fishes was 2.0 to 7.8 cm and the size range for non-indigenous fishes was 1.7 to 8.6 cm. Median parasite intensity was significantly higher for indigenous fishes (8) compared with non-indigenous fishes (0) (U = 166569.0; $p < 0.001$). For all of the non-indigenous species tested, there was a moderate to weak correlation between host size and parasite intensity (Table 6.4). For all species of indigenous fish tested, except the Purple-spotted gudgeon, there was a strong correlation between host size and parasite intensity (Table 6.4)

Table 6.3 Summary statistics for parasite communities of indigenous and non-indigenous host fishes from northern Queensland fresh waters (∑n = number of fish examined; Mean spp = mean number of parasite species per host; Max spp. = maximum number of parasite species recorded per host; ∑ spp prev. = total parasite prevalence per host species; dlc:ilc = ratio of direct life cycle to indirect life cycle parasites; r = correlation coefficient; a = predicted maximum component parasite community richness; b = index of mean parasite infracommunity richness G = parasite community interactivity index; r^2 = coefficient of determination

						HOST SIZE and PARASITE INTENSITY				**INTERACTIVITY**			
Host	**∑n**	**Mean spp.**	**Max.spp**	**∑ spp prev.**	**dlc:ilc**	**∑infect. fish**	**r**	**p**	**association**	**a**	**b**	**G**	**r^2**
NONINDIG.													
MM	304	0.45	2	39.8		121	0.087	0.344	v. weak	9.54	0.016	0.153	0.964
MM*(adult)	332									-	-	-	-
HG	10	0.30	1	30.0		-	-	-	-	-	-	-	-
HB	22	0.32	1	31.8		-	-	-	-	-	-	-	-
GAM	711	0.21	2	19.1		136	0.194	0.023	v. weak	6.55	0.0075	0.049	0.934
PR	514	0.33	3	29.0		149	0.226	0.006	weak	8.48	0.011	0.093	0.940
XM	338	0.39	2	36.4		123	0.260	0.004	weak	4.48	0.011	0.049	0.760
XH	25	0.45	2	32.0		8	-0.229	0.585	weak	-	-	-	-
TT	25	0.00	0	0.0	-	-	-	-	-	-	-	-	-
∑spp	1949	0.32	3	28.1	0.75	547	0.072	0.091	v. weak	-	-	-	-
INDIG.													
RF	174	2.60	7	96.0		174	0.723	<0.001	strong	11.30	0.075	0.850	0.817
GP	191	1.62	5	91.6		174	0.739	<0.001	strong	7.49	0.051	0.382	0.919
HH	168	2.56	5	97.6		164	0.757	<0.001	strong	8.09	0.236	1.910	0.623
EG	156	0.93	2	83.3		131	0.560	<0.001	mod. strong	7.97	0.022	0.175	0.977
PSG	32	0.75	3	53.1		17	0.287	<0.001	weak	5.03	0.152	0.765	0.999
MA	101	1.29	6	87.1		92	0.626	<0.001	mod. strong	10.03	0.088	0.910	0.940
∑spp	822	1.81	7	90.0	0.88	752	0.598	<0.001	mod. strong	-	-	-	-

Indigenous fish: GP = Agassiz's glassperch; RF = Eastern Queensland rainbowfish; HH = Fly-specked hardyhead; MA = Mouth almighty; EG = Empire gudgeon; PSG = Purple-spotted gudgeon. Non-indigenous fish: MM = Mozambique mouthbrooder; HG = Jewel cichlid; HB = Burton's haplochromis; GAM = Mosquitofish; PR = Guppy; XM = Platy; XH = Swordtail; TT = Three-spot-gourami



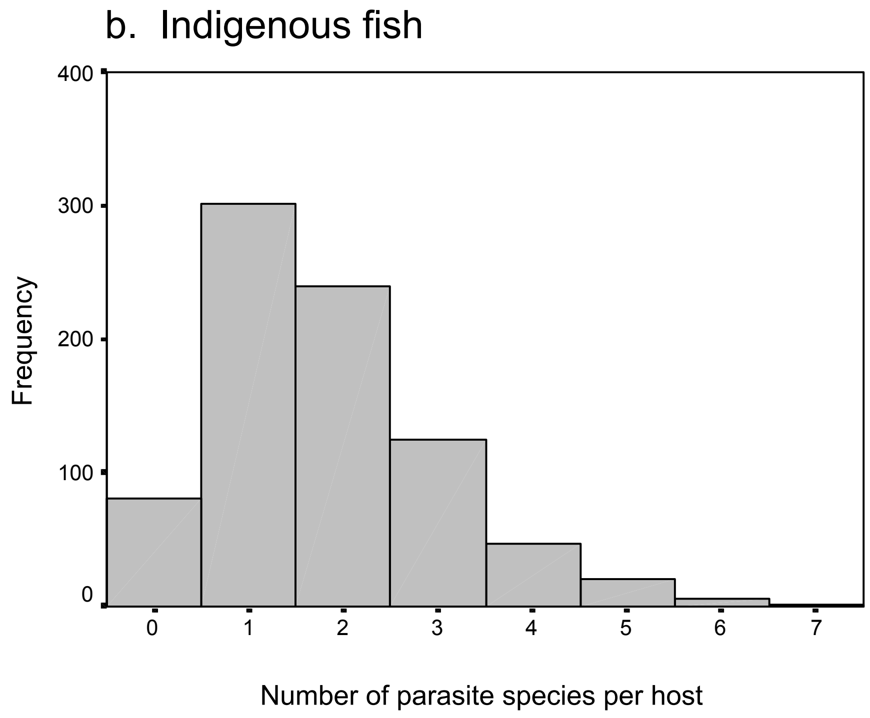


Figure 6.6 Regional parasite species richness frequency distributions for freshwater fishes in northern Queensland: a, non-indigenous fish, b, indigenous fish

Table 6.4 Correlation between parasite intensity and host length for samples of indigenous and non-indigenous freshwater fish species from northern Queensland

NON-INDIGENOUS FISH	**r**	**p**	**Sample size**
Guppy	0.226	0.006	149
Mosquitofish	0.087	0.344	121
Swordtail	0.229	0.585	8
Platy	0.260	0.004	123
Mozambique mouthbrooder	0.087	0.344	121
INDIGENOUS FISH			
Empire gudgeon	0.560	<0.001	131
Purple-spotted gudgeon	0.287	0.265	17
Fly-specked hardyhead	0.757	<0.001	164
Agassiz's glassperch	0.739	<0.001	174
Eastern Queensland rainbowfish	0.723	<0.001	175
Mouth almighty	0.626	<0.001	92

Core-satellite species

The regional parasite community of non-indigenous fishes had a unimodal structure (Figure 6.7) with no core taxa and total prevalences for all parasite taxa less than 30% for all species examined (Figure 6.8). On only two occasions were site sample prevalences for non-indigenous fishes greater than 70% (the non-indigenous monogenean, *Cichlidogyrus tilapiae,* on Mozambique mouthbrooder, from Upper Cranbrook Creek, Townsville (70.1%), and a native spirurid nematode (sp. 1) on Mosquitofish from Majors Creek (76.6%). For non-indigenous fish, monogeneans and nematodes were the dominant parasite taxa with 42.6% and 35.2% of the total parasites recorded. All of the four monogenean species were non-indigenous and introduced with the hosts, while the nematodes were indigenous species. Of the latter, the small larval spirurid (sp. 1) with low host specificity provided the largest proportion of total numbers. In contrast, regional parasite community structure for indigenous fish hosts had a bimodal prevalence distribution (Figure 6.7) with at least one core and several intermediate and satellite taxa for the majority of species examined (Figure 6.9).

Community interactivity

The NMDS ordination plot (Figure 6.10) separated the regional parasite faunas of non-indigenous and indigenous host fishes into two clear groups: parasite communities of non-indigenous hosts were more similar to each other than they were to indigenous host assemblages.

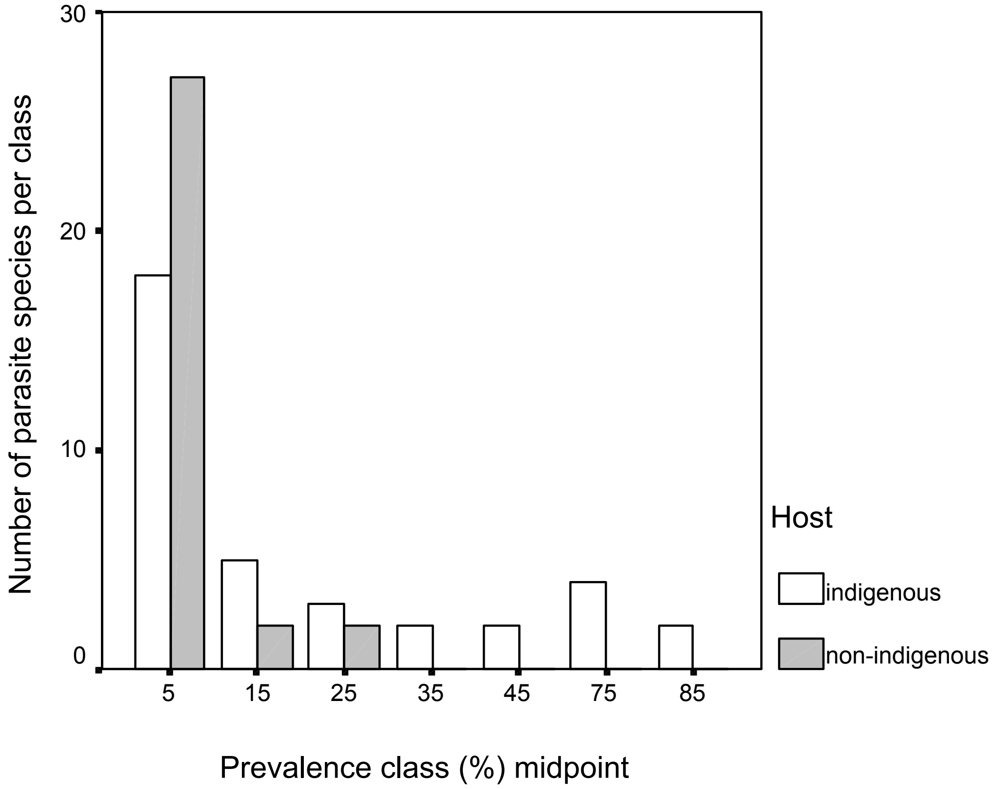
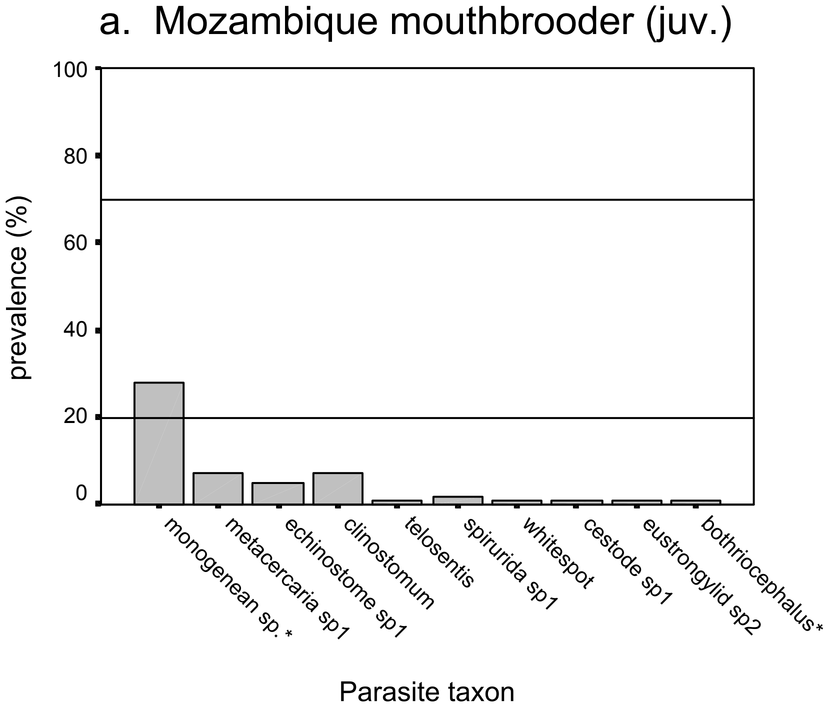


Figure 6.7 Regional parasite community structure (core and satellite taxa prevalences) for non-indigenous and indigenous freshwater fishes in northern Queensland



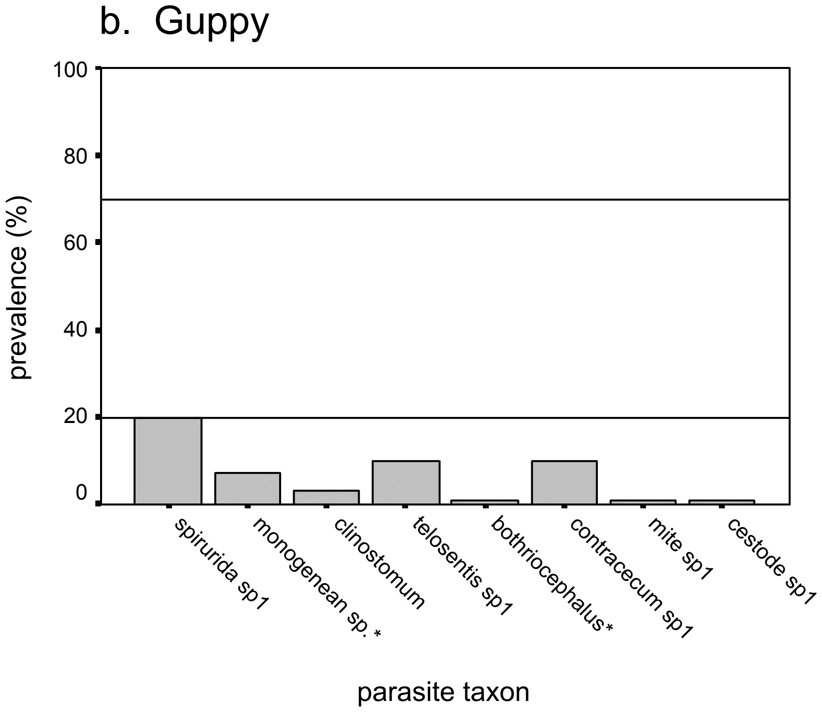
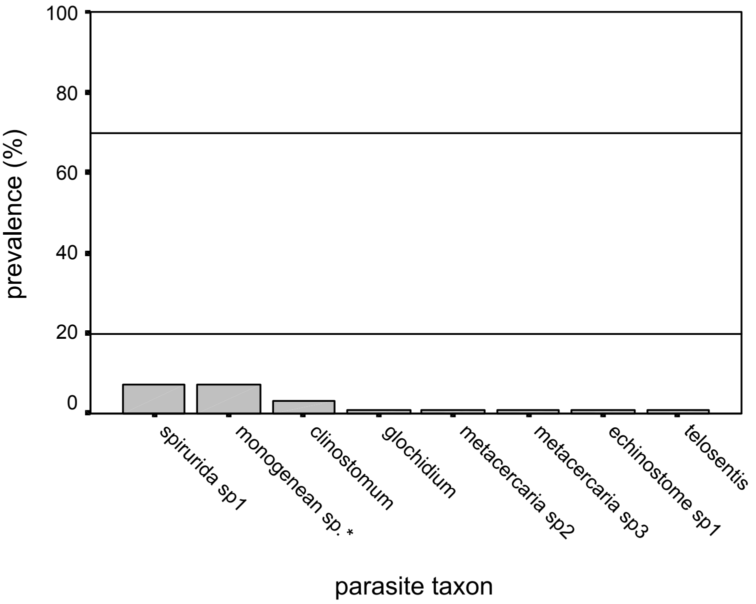


Figure 6.8 Prevalence of parasite taxa for non-indigenous freshwater fish species in northern Queensland: a, Mozambique mouthbrooder (juv.), b, Guppy, c, Mosquitofish, d, Platy

continued

c. Mosquitofish



d. Platy

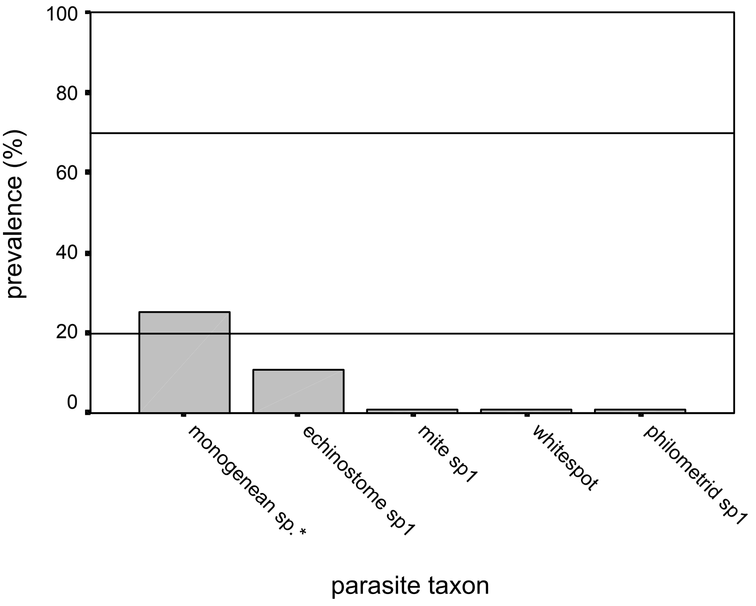


Figure 6.8 continued

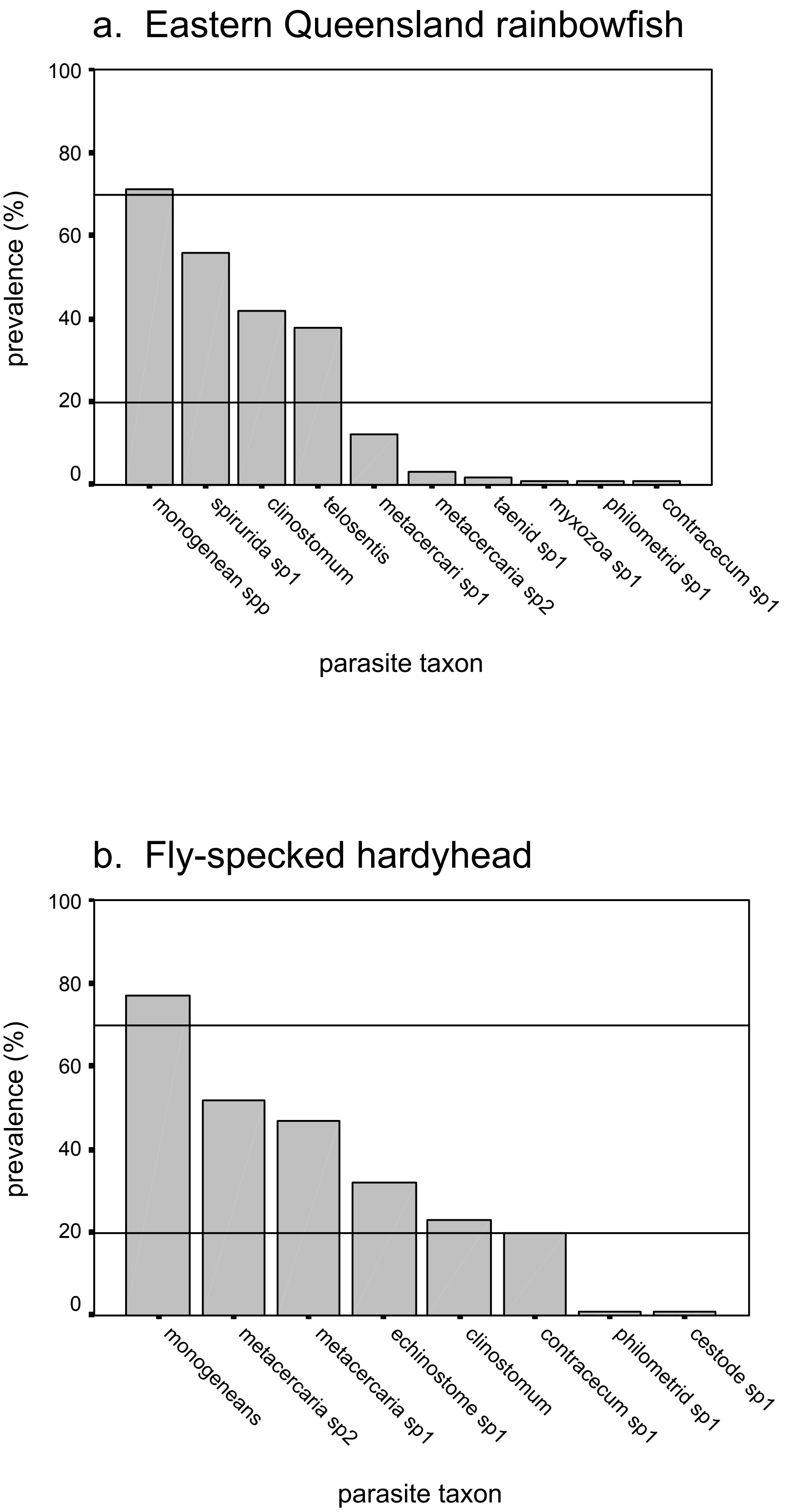
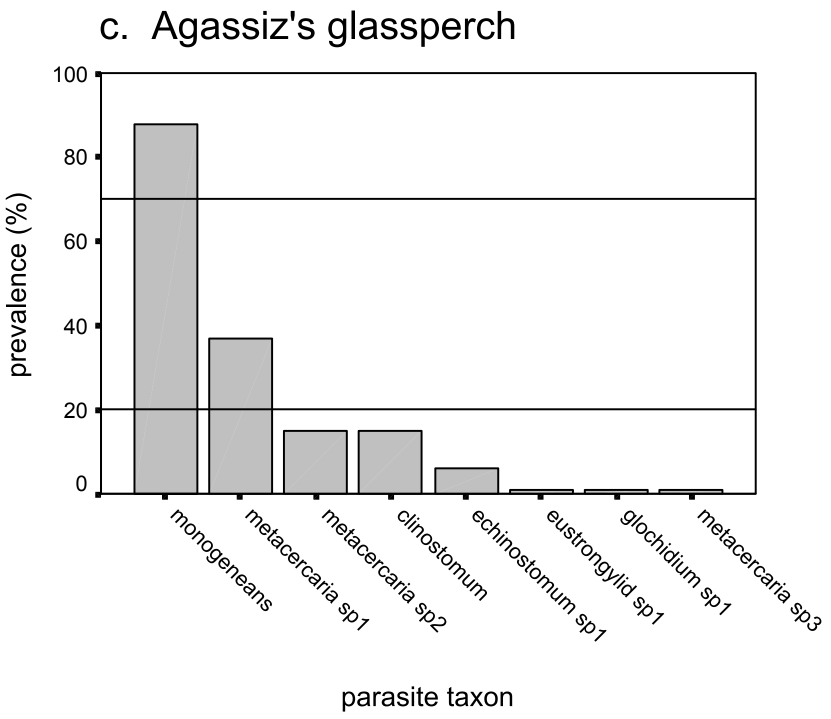


Figure 6.9 Prevalence of parasite taxa for indigenous freshwater fish species in northern Queensland: a, Eastern Queensland rainbowfish, b, Fly-specked hardyhead, c, Agassiz's glassperch, d, Empire gudgeon, e, Mouth almighty (juv.) continued



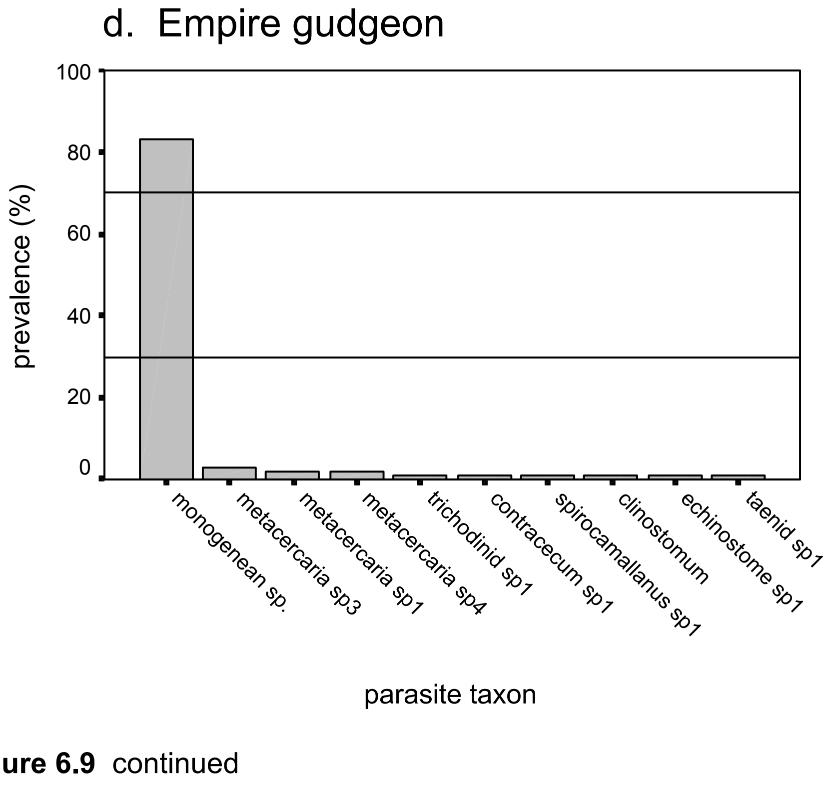


Figure 6.9 continued

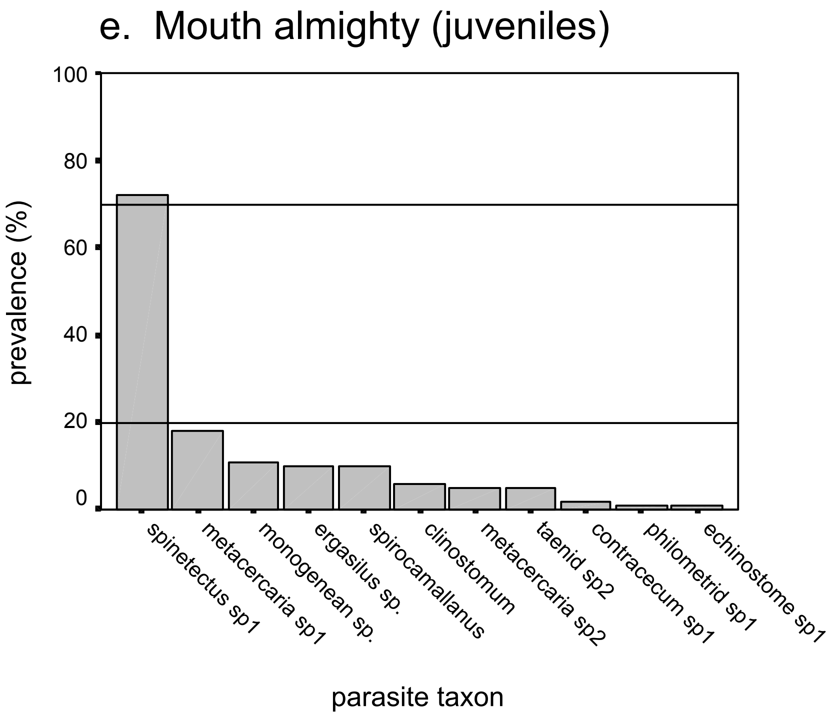


Figure 6.9 continued

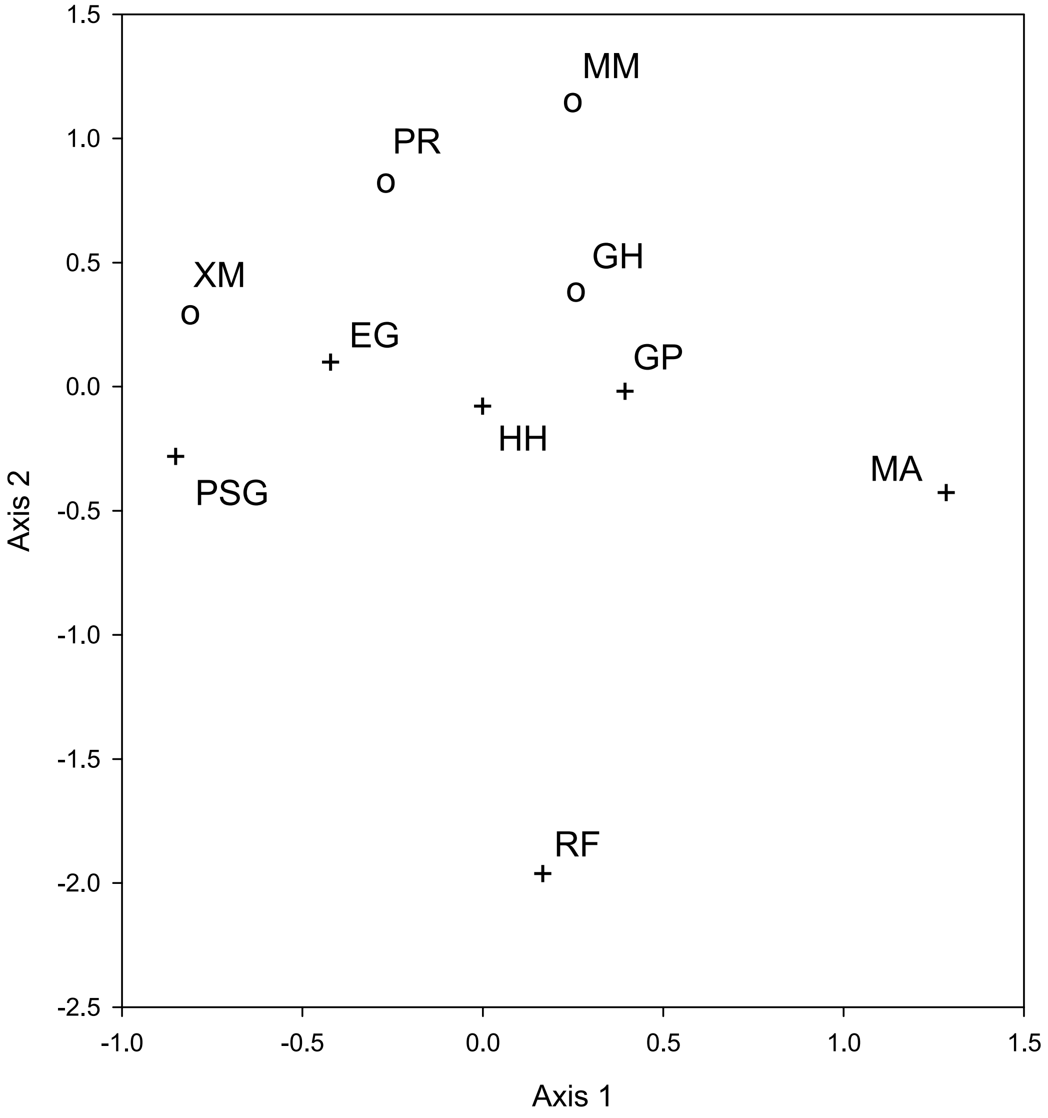


Figure 6.10 Two-dimensional (NMDS) ordination plot for comparison of indigenous and non-indigenous fish host parasite faunas based on parasite presence/absence

(o) non-indigenous host: MM = Mozambique mouthbrooder; PR = Guppy; XM = Platy; GH = Mosquitofish;

(+) indigenous host: EG = Empire gudgeon; GP = Agassiz's glassperch; HH = Fly-specked hardyhead; PSG = Purple-spotted gudgeon; MA = Mouth almighty; RF = Eastern Queensland rainbowfish

The degree of sharing of parasites was low: of the 45 parasite species recorded, from both non-indigenous and indigenous hosts, 15 species of indigenous parasites were found on non-indigenous hosts (33%) (Tables 6.1 and 6.2). Of these only 10 species were observed on both non-indigenous and indigenous hosts during the study. Since the shared parasites were most probably indigenous species, the direction of transfer was probably from indigenous to non-indigenous hosts.

A summary of the interactivity indices is presented in Table 6.3. Cumulative species curves (Figure 6.11) showed a good fit to the two-parameter Weibull curve with high r^2 values. The curves for indigenous fishes were steeper than those for non- indigenous fishes. Maximum predicted regional richness values (*a*) for indigenous fish were very probably underestimates due to relatively small sample sizes.

The mean value for indigenous fishes (8.12) was higher than that for non-indigenous fishes (7.26) but the difference was not significant ($t = 0.661$; $df = 8$; $p = 0.549$). The highest values were for the Eastern Queensland rainbowfish and Mouth almighty, while the Mozambique mouthbrooder and Guppy had the third and fourth highest values respectively (Table 6.3). The index of mean infracommunity richness (*b*) values were all higher for indigenous species (overall mean: 0.1040) compared with non-indigenous species (0.0114), and the interactivity index G values were also all higher for indigenous fish species (overall mean: 0.832) compared with non-indigenous fish species (0.086).

Parasite acquisition and non-indigenous host residency time

A plot of the number of indigenous parasites acquired against an estimate of residency time for several non-indigenous fish hosts in northern Queensland (13-20° S) is shown in Figure 6.12. Plots of parasite acquisition by non-indigenous hosts in southern Australia and Great Britain are presented in Appendix E, Figures E1 and E2 for comparison.

There was a strong positive, significant correlation between parasites acquired and residence time for non-indigenous fishes in fresh waters in tropical northern Queensland. The r^2 value for the fitted regression line was 0.702, i.e., about 70 percent of the variation in the data can be accounted for by residence time (Table 6.5). For a generalised non-indigenous fish species to acquire a parasite fauna of between 15 and 25 species (similar to that of common indigenous hosts such as Agassiz's glassperch, Eastern Queensland rainbowfish and Empire gudgeon (see Table 6.2 and Dove (1998)), the residence time required was estimated by extrapolation to be between 109 and 187 years (Table 6.5).

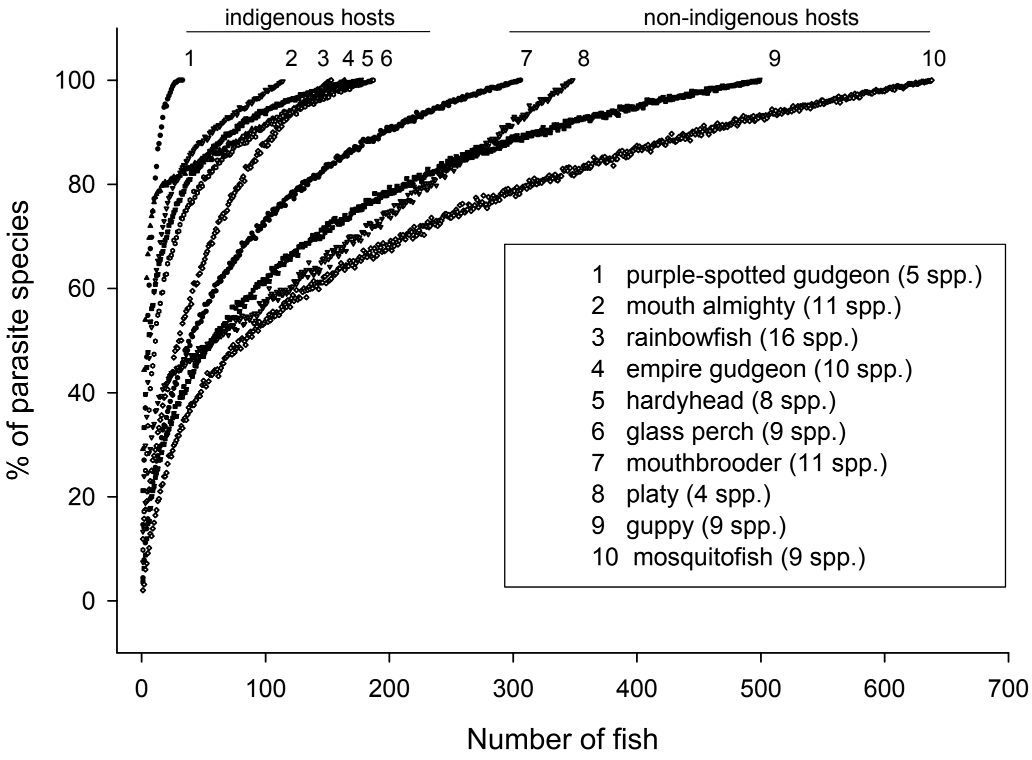


Figure 6.11 Cumulative species curves for regional parasite communities of non-indigenous and indigenous freshwater fishes in northern Queensland (total parasite richness per host species in parentheses)

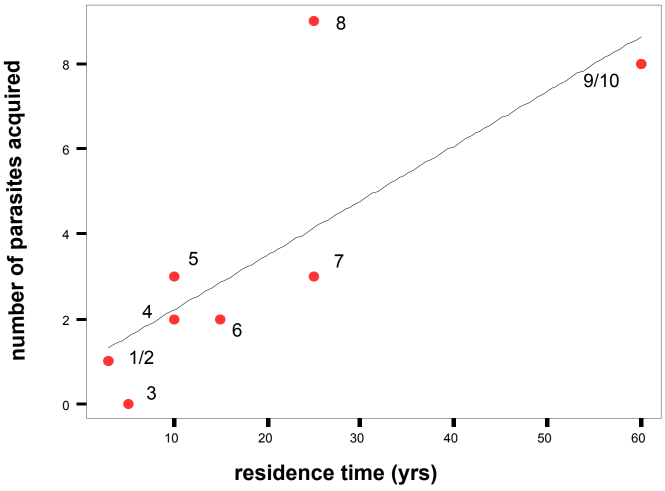


Figure 6.12 Parasites acquired by non-indigenous fishes in relation to residence time in tropical northern Queensland fresh waters. Hosts: 1 = Burton's haplochromis; 2 = Jewel cichlid; 3 = Three-spot gourami;; 4 = Oscar; 5 = Midas cichlid; 6 = Black-spotted mangrove cichlid; 7 = Platy; 8 = Mozambique mouthbrooder; 9 = Guppy; 10 = Mosquitofish

Correlation and regression summaries are presented in Table 6.5 for southern Australian and Great Britain data for comparison with the northern Queensland data. There was a strong, positive, significant correlation for both southern Australia and Great Britain between parasite acquisition and residence time of non-indigenous fish species. There was also a positive relationship between increasing latitude and increasing parasite acquisition time. The acquisition rate for southern Australian hosts was about four times slower, while the acquisition rate for hosts in Great Britain was about 10 times slower than the rate for tropical northern Queensland hosts.

Table 6.5 Correlation and Regression analyses summary for parasite acquisition in relation to residence time of non-indigenous freshwater fishes

Location	Latitude	Number of host species	r	p	r^2	Regression slope (a)	Residence time (years)
Northern QLD	13-20ºS	11	0.838	0.002*	0.702	0.923	109-187
Southern Australia	30-45ºS	6	0.859	0.028*	0.738	0.0321	462-774
Great Britain	50-58ºN	7	0.750	0.050*	0.570	0.01465	1049 - 1731

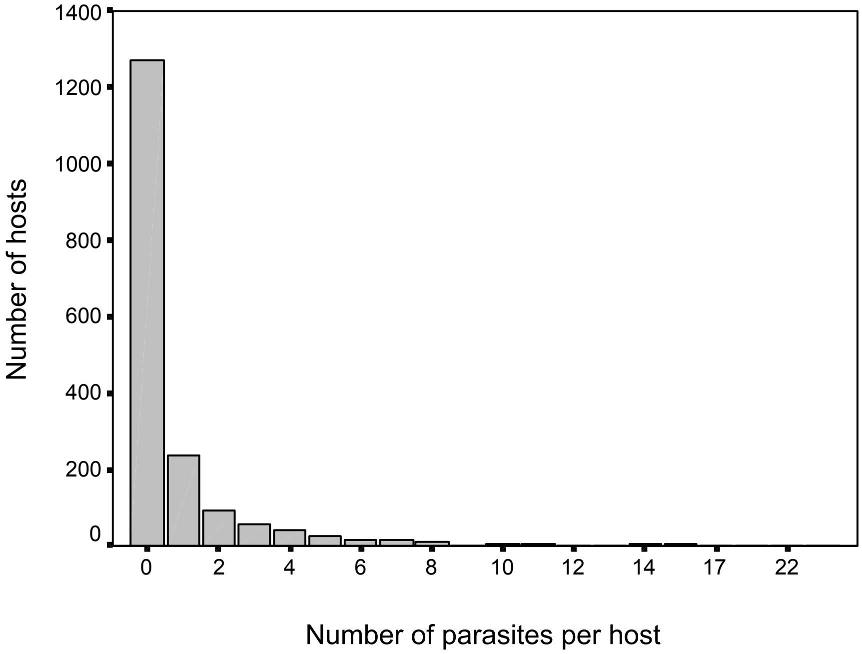
Using the regression equation obtained for non-indigenous hosts in Great Britain, by extrapolation, the residency estimate for a Brown trout to acquire at least 50 parasites was 3438 years, well below the species' known residency time in Britain of about 10, 000 years. While the period prior to host saturation may be described by a linear model, the longer term relationship between parasite acquisition and host residency, at least for temperate fresh waters, might be non-linear.

Host-parasite frequency distributions

The total parasite load frequency distribution for non-indigenous hosts (Figure 6.13a), and for individual hosts fitted the negative binomial model (Table 6.6), while the frequency distribution for indigenous hosts (Figure 6.13b) differed significantly from the negative binomial model (Table 6.7).

For the six non-indigenous fish species tested (Mozambique mouthbrooder, Burton's haplochromis, Mosquitofish, Guppy, Platy and Swordtail), the parasite frequency distributions for all parasite taxa combined fitted the negative binomial distribution. (Table 6.6 and Figure 6.14). Twelve out of 13 separate parasite taxa frequency distributions tested for non-indigenous

a. Non-indigenous fishes



b. Indigenous fishes

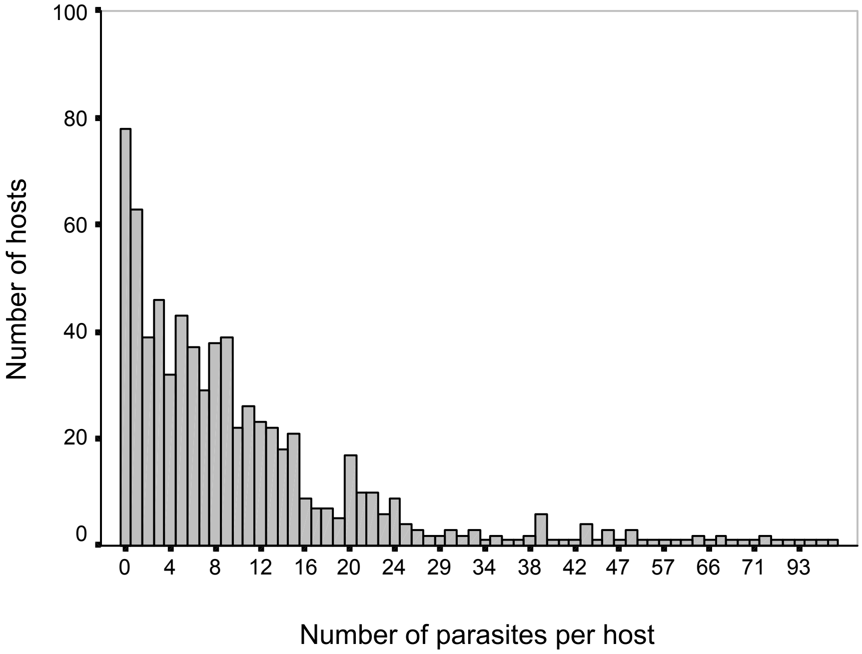


Figure 6.13 Parasite frequency distributions for freshwater fishes in northern Queensland: a, non-indigenous fishes, b, indigenous fishes

Table 6.6 Summary statistics for parasite frequency distribution on hosts and goodness of fit to the negative binomial model for non-indigenous fish species collected in fresh waters in northern Queensland

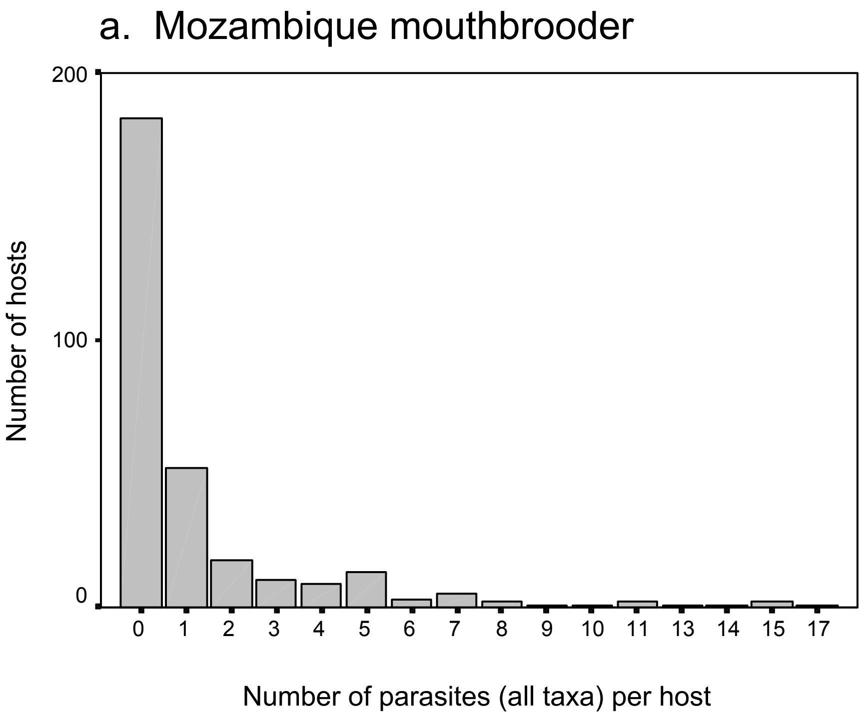
Host	Parasite Taxon	mean	variance	*k*	χ^2	df	p
MM	All taxa	1.33	7.12	0.3055	11.506	10	0.320
	Monogenea	0.91	5.43	0.1832	5.302	10	0.870
	Metac. cyst (liver)	0.27	1.61	0.0544	10.359	6	0.110
	Echinostome	0.09	0.19	0.0866	10.363	3	0.016
HB	All taxa	0.77	1.99	0.4860	3.876	3	0.275
GAM	All taxa	0.48	2.40	0.1200	7.246	9	0.612
	Monogenea	0.14	0.44	0.0653	3.083	5	0.687
	Nematoda	0.27	1.92	0.0442	10.627	7	0.156
	Clinostomum	0.03	0.03	0.5491	0.001	2	1.000
PR	All taxa	0.97	6.45	0.1717	3.344	9	0.949
	Monogenea	0.16	0.69	0.0483	3.312	6	0.769
	Nematoda	0.68	5.21	0.1021	5.072	9	0.828
	Clinostomum	0.08	0.35	0.0265	3.671	4	0.452
XM	All taxa	0.78	1.92	0.5337	4.852	8	0.773
	Monogenea	0.51	1.36	0.306	2.914	6	0.820
	Echinostome	0.25	0.72	0.1330	7.224	5	0.205
XH	All taxa	0.76	2.11	0.4278	0.744	3	0.863
	Whitespot	0.52	1.26	0.3654	0.279	3	0.964
Σ SPP	All taxa	0.81	4.21	0.1930	21.374	13	0.066

Hosts: MM = Mozambique mouthbrooder; HB = Burton's haplochromis; GAM = Mosquitofish; PR = Guppy; XM = Platy; XH = Swordtail

Table 6.7 Summary statistics for parasite frequency distribution on hosts and goodness of fit to the negative binomial model for indigenous fish species collected in fresh water in northern Queensland

Host	Parasite Taxon	mean	variance	*k*	χ^2	df	p
RF	All taxa	13.21	270.09	0.6793	55.077	22	<0.001
	Monogenea	4.97	59.64	0.4338	13.433	16	0.641
	Nematoda	4.77	103.66	0.2301	20.837	15	0.142
	Clinostomum	1.28	5.98	0.3486	5.841	8	0.665
	Metac. cyst (a)	0.47	2.77	0.0960	8.284	6	0.218
	Acanthocephala	1.30	6.72	0.3130	6.982	10	0.727
GP	All taxa	15.72	228.10	1.1636	41.111	26	0.030
	Monogenea	11.99	141.87	1.1069	30.685	23	0.131
	Clinostomum	0.27	0.67	0.1822	0.656	5	0.985
	Metac. cyst (a)	0.63	1.19	0.7087	8.348	5	0.138
	Metac. cyst (b)	2.15	94.05	0.0503	25.585	7	0.001
	Echinostome	0.28	2.09	0.0433	2.367	3	0.500
HH	All taxa	12.82	199.09	0.8823	49.181	23	0.001
	Monogenea	5.89	79.55	0.4710	16.191	17	0.510
	Nematoda	0.31	0.49	0.5339	4.190	3	0.242
	Clinostomum	1.12	7.48	0.1972	13.442	9	0.144
	Metac. cyst (a)	1.77	11.21	0.3319	13.276	8	0.103
	Metac. cyst (b)	2.03	10.04	0.5145	4.558	12	0.971
	Echinostome	1.63	28.86	0.0976	15.726	9	0.073
EG	All taxa	9.24	151.24	0.6012	24.507	22	0.321
	Monogenea	8.27	142.10	0.5110	34.884	22	0.040
PSG	All taxa	1.69	5.13	0.8303	1.326	4	0.857
	Monogenea	0.58	2.82	0.1502	0.897	3	0.826
	Clinostomum	0.47	1.28	0.2727	0.132	3	0.988
MA.juv	All taxa	8.69	123.37	0.6585	18.431	14	0.188
	Monogenea	0.19	0.39	0.0902	0.681	4	0.954
	Copepoda	0.31	1.37		5.009	4	0.286
	Nematoda	5.42	28.80	1.2565	12.395	15	0.649
	Metac. cyst (a)	2.10	71.71		1.286	4	0.864
ΣSPP	All taxa	11.93	205.38	0.7357	61.863	34	0.002

Hosts: RF = Eastern Queensland rainbowfish; GP = Agassiz's glassperch; HH = Fly-specked hardyhead; EG = Empire gudgeon; PSG = Purple-spotted gudgeon; MA.juv. = Mouth almighty (juv.)



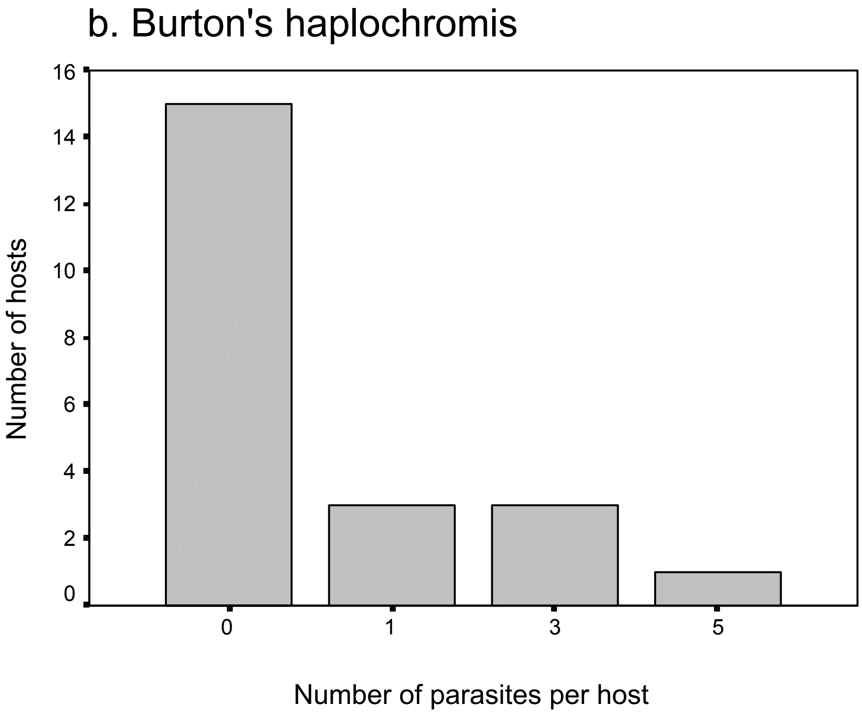


Figure 6.14 Parasite frequency distribution (all taxa) per host for non-indigenous freshwater fishes in northern Queensland: a, Mozambique mouthbrooder, b, Burton's haplochromis, c, Mosquitofish, d, Guppy, e, Platy, f, Swordtail

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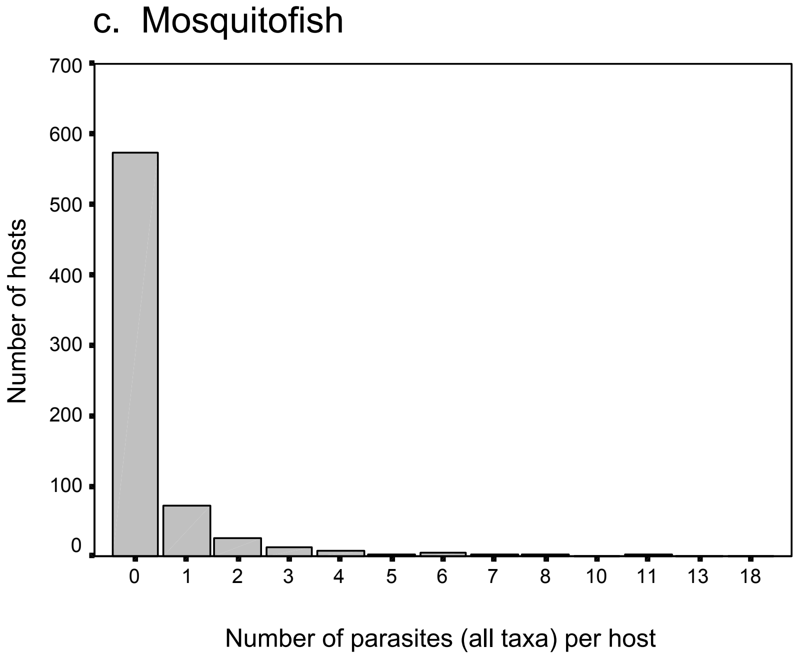

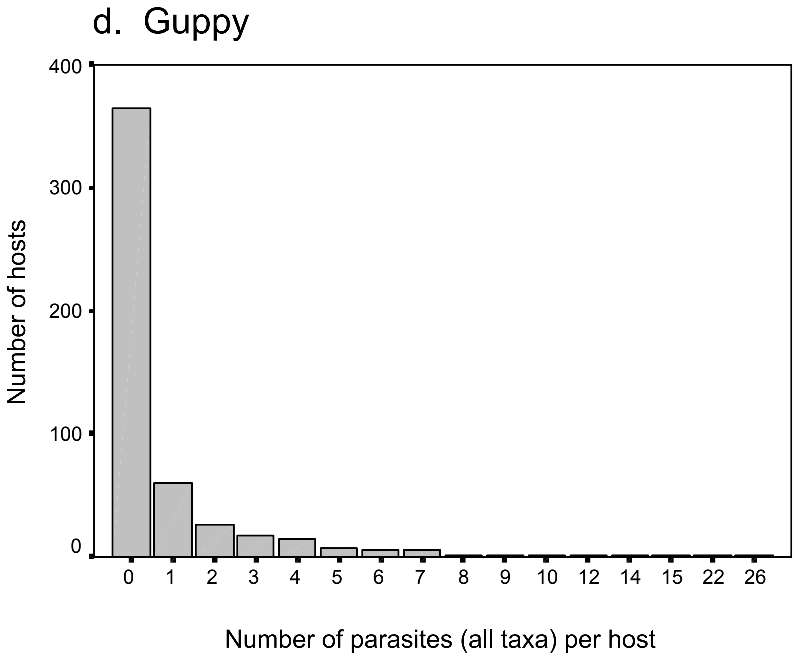

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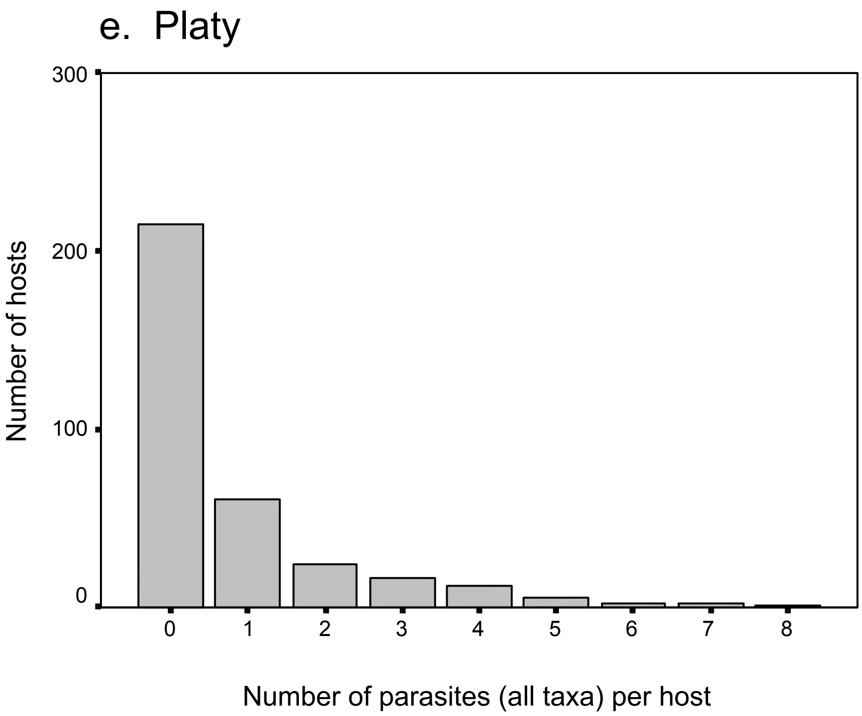


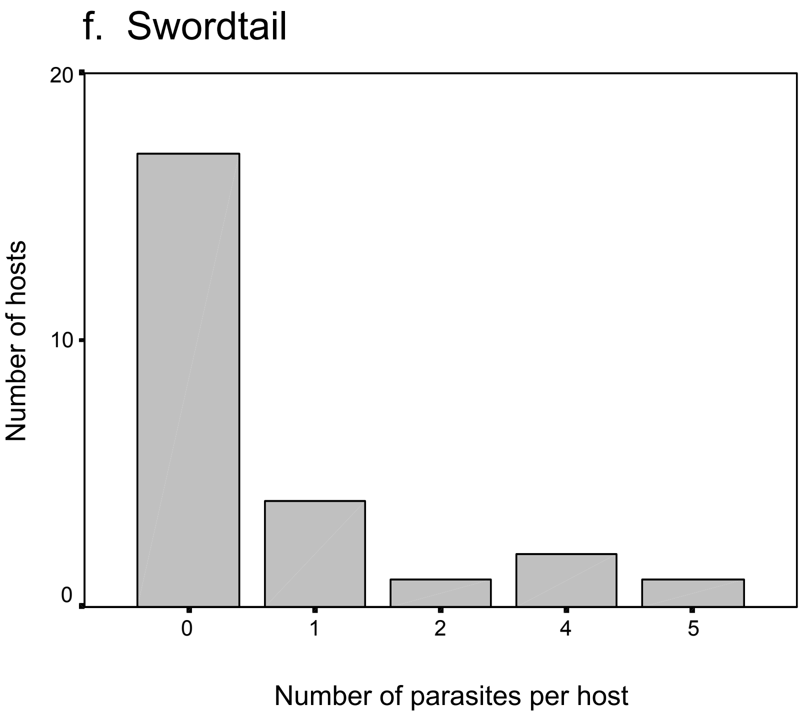


Figure 6.14 continued

fish hosts fitted the negative binomial model, while one distribution closely approximated it. (Table 6.6).

For the six indigenous fish species tested (Eastern Queensland rainbowfish, Fly-specked hardyhead, Agassiz's glassperch, Empire gudgeon, Purple-spotted gudgeon and Mouth almighty juveniles), three of the six frequency distributions (Eastern Queensland rainbowfish, Agassiz's glassperch and Fly-specked hardyhead) for all parasite taxa combined differed significantly from the negative binomial model with multi-modal distributions (Table 6.7 and Figures 6.15). For 23 separate parasite taxa tested for indigenous fish hosts, 21 frequency distributions fitted the negative binomial model and the remaining two approximated closely to it (Table 6.7). The median *k* parameter for the negative binomial distribution for individual and total parasite taxa frequency distributions from non–indigenous fishes (0.0943; 0.3666) (Table 6.6) were significantly lower than the corresponding values (Table 6.7) for indigenous fishes (0.2727; 0.7548) (U = 80.00; p = 0.044; U = 0.000; p = 0.004). Parasite distributions for non-indigenous fishes were more aggregated or over-dispersed compared with distributions for indigenous fishes.

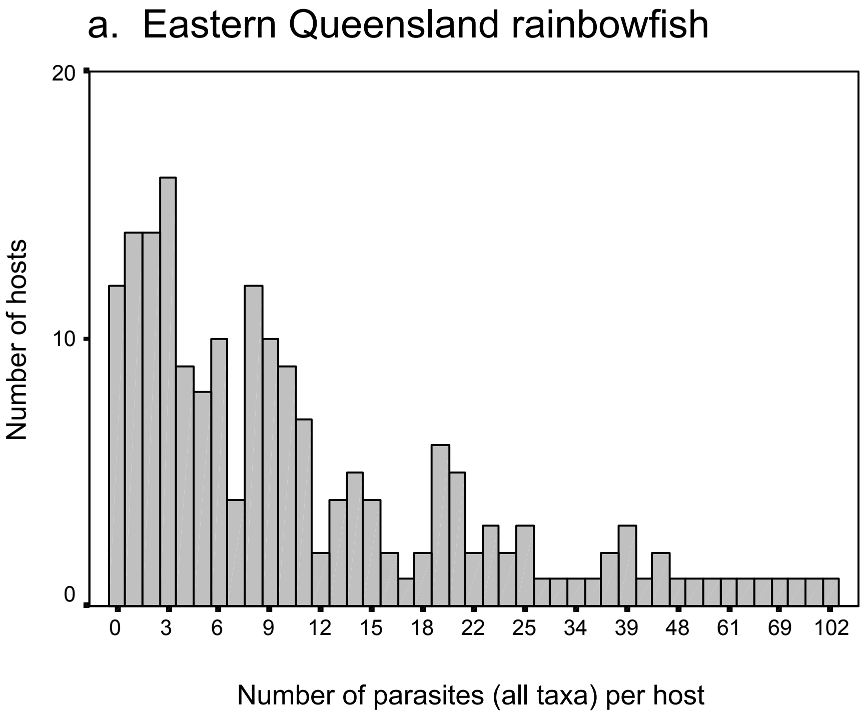
6.3.3 Parasite species richness, aggregation and habitat disturbance

Parasite species richness

Parasite species richness was significantly lower for non-indigenous fishes in high disturbance habitats (U= 20.50; p = <0.001) and low disturbance habitats (U = 19.5; p = 0.044) compared with indigenous fishes. In high disturbance habitats, median richness for non-indigenous fishes was 0.5 compared with 4.5 for indigenous fishes. In low disturbance habitats, median richness for non-indigenous hosts was 2 compared with 3 for indigenous hosts.

Parasite aggregation in relation to intensity and prevalence for non-indigenous and indigenous fishes in high and low disturbance habitats

There was a strong positive correlation between intensity and prevalence for non-indigenous hosts (0.838) and a moderate correlation for indigenous hosts (0.599) in high disturbance habitats, and a strong correlation for both indigenous and non-indigenous hosts (0.903 and 0.871) in low disturbance habitats (Figure 6.16). For non-indigenous fishes, change in prevalence was greater than change in intensity, while for indigenous fishes, there was a greater change in intensity than change in prevalence in both low and high disturbance habitats



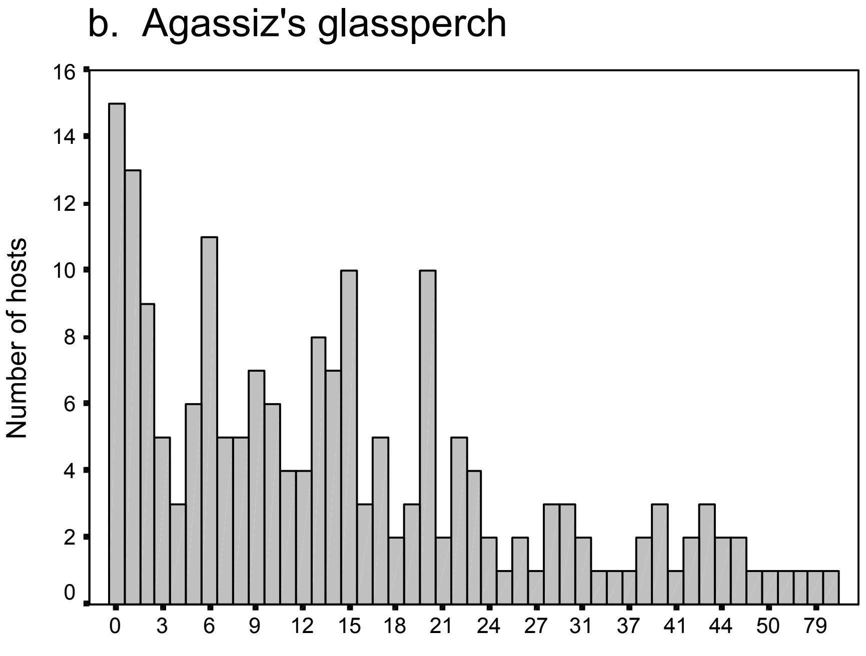
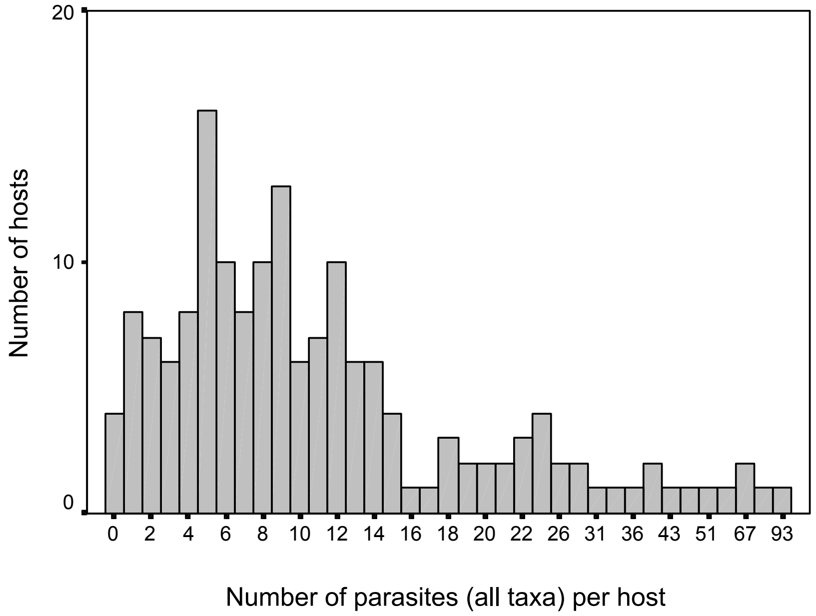


Figure 6.15 Parasite frequency distributions (all taxa) per host for indigenous freshwater fish species in northern Queensland: a, Eastern Queensland rainbowfish, b, Agassiz's glassperch, c,Fly-specked hardyhead, d, Empire gudgeon, e, Purple-spotted gudgeon, f, Mouth almighty

continued

c. Fly-specked hardyhead



d. Empire gudgeon

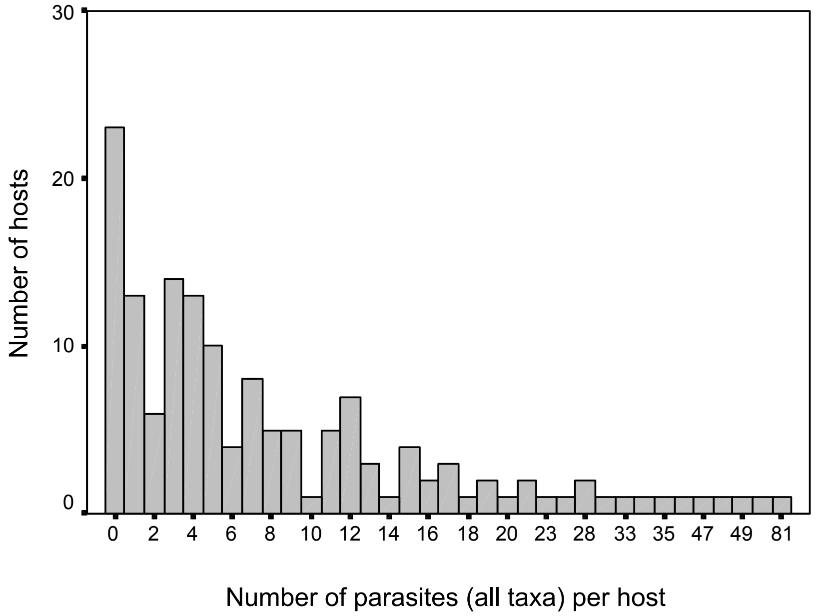
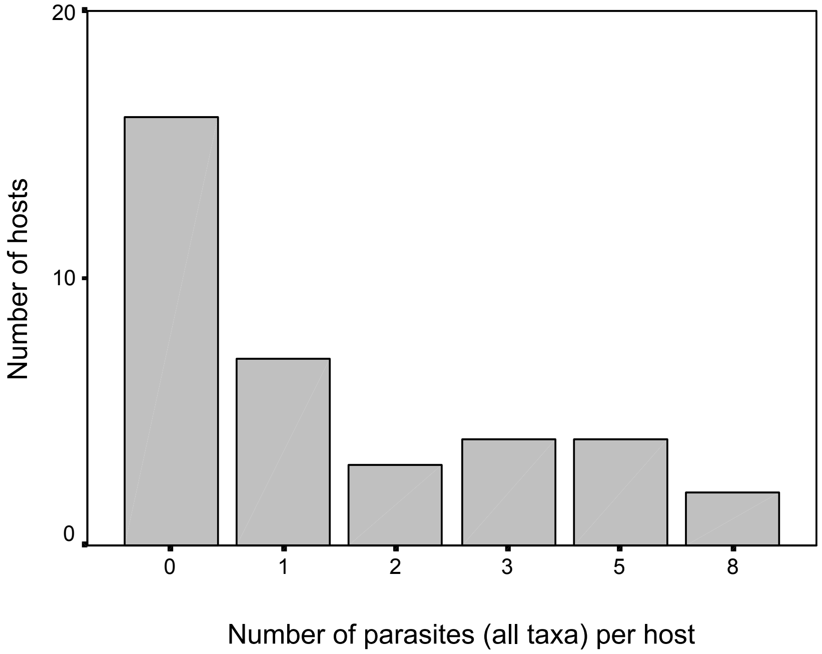


Figure 6.15 continued

e. Purple-spotted gudgeon



f. Mouth almighty

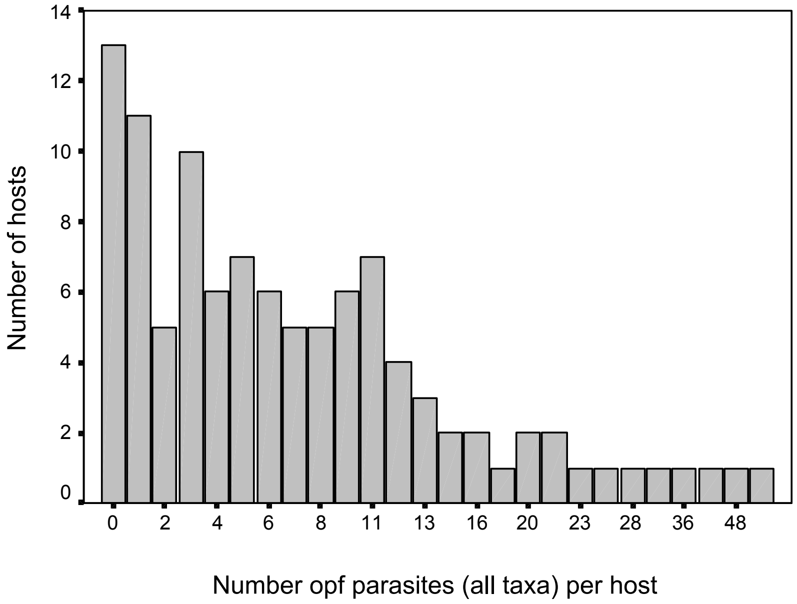


Figure 6.15 continued

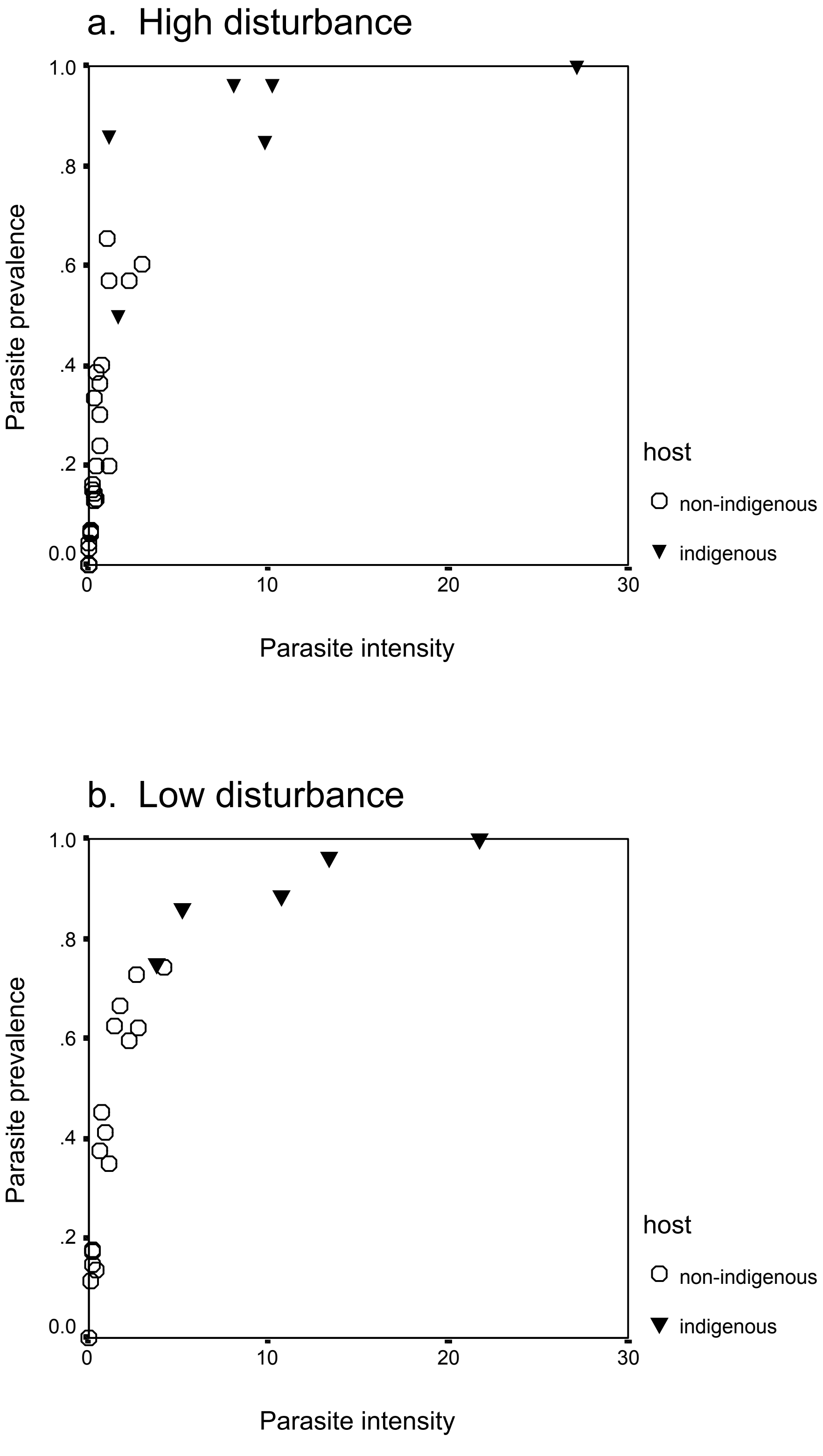


Figure 6.16 Change in parasite prevalence with intensity for non-indigenous and indigenous freshwater fish hosts: a, high disturbance habitats, b, low disturbance habitats

For combined hosts, the prevalence/intensity relationship was clearly curvilinear irrespective of disturbance. Non-indigenous fishes were located along the “unsaturated” end of the curve (low intensity and prevalence) while indigenous fishes were located toward the "saturated" end of the curve with much higher intensity and prevalence values than non-indigenous fishes. In high disturbance habitats, there were moderate to strong significant correlations for parasite intensity and prevalence in relation to k (aggregation) parameter in high and low disturbance habitats for indigenous and non-indigenous fish hosts (Table 6.8 and Figure 6.17 and 6.18). As intensity and prevalence decreased the aggregation parameter k decreased, i.e., parasites became more aggregated on hosts. For non-indigenous fishes, the correlation values for prevalence and intensity in relation to the k parameter were also higher in high disturbance habitats compared with low disturbance habitats and also in comparison with indigenous fishes in high disturbance and low disturbance habitats.

Intensity and prevalence were significantly lower for non-indigenous fish hosts in high disturbance habitats compared with low disturbance habitats, although the difference for indigenous hosts was not significant. Parasites were significantly more aggregated (lower *k*) for non-indigenous hosts in high disturbance than low disturbance habitats, while the difference for indigenous hosts was not significant. In both high disturbance and low disturbance habitats, non-indigenous fishes had significantly lower prevalence and intensity than indigenous fishes. Parasites were significantly more aggregated on non-indigenous hosts than indigenous hosts in high disturbance habitats, although there was no significant difference in *k* for indigenous and non-indigenous fishes in low disturbance habitats (Table 6.9).

Table 6.8 Correlation summary for parasite frequency distribution *k* (aggregation) parameter in relation to prevalence and intensity for indigenous and non-indigenous freshwater fish hosts in low and high disturbance habitats (n = sample size; r = Pearson correlation coefficient; p = probability; disturb. = level of habitat disturbance)

disturbance					***k v* prevalence**				
	indigenous + non-indigenous hosts			**Indigenous hosts**			**Non-indigenous hosts**		
	n	r	p	n	r	p	n	r	p
LOW	21	0.375	0.094	5	0.346	0.568	16	0.440	0.088
HIGH	48	0.795	<0.001	6	0.482	0.332	42	0.637	<0.001
					***k v* intensity**				
	n	r	p	n	r	p	n	r	p
LOW	21	0.099	0.671	5	0.084	0.894	16	0.186	0.490
HIGH	48	0.649	0.001	6	0.473	0.344	42	0.479	0.001

Table 6.9 Statistical summary for comparison of parasite intensity, prevalence and aggregation (*k*) of regional parasite communities for indigenous and non-indigenous freshwater fish hosts in high and low disturbance habitats in northern Queensland

	Mann-Whitney	**Intensity**	**Prevalence**	***k***
Non-indigenous hosts	U	133.50	127.00	117.00
	p	<0.001	<0.001	<0.001
HIGH disturbance	(median)	0.036	0.038	0.0000
LOW disturbance	(median)	0.795	0.393	0.6489
Indigenous hosts	U	11.00	13.50	14.00
	p	0.465	0.783	0.855
HIGH disturbance	(median)	8.90	0.912	1.1929
LOW disturbance	(median)	10.69	0.886	1.2308
HIGH (indig./non-indig.)	U	1.00	0.00	20.00
	p	0.001	0.001	<0.001
LOW (indig./non-indig.)	U	6.00	4.00	33.00
	p	<0.001	<0.001	0.563

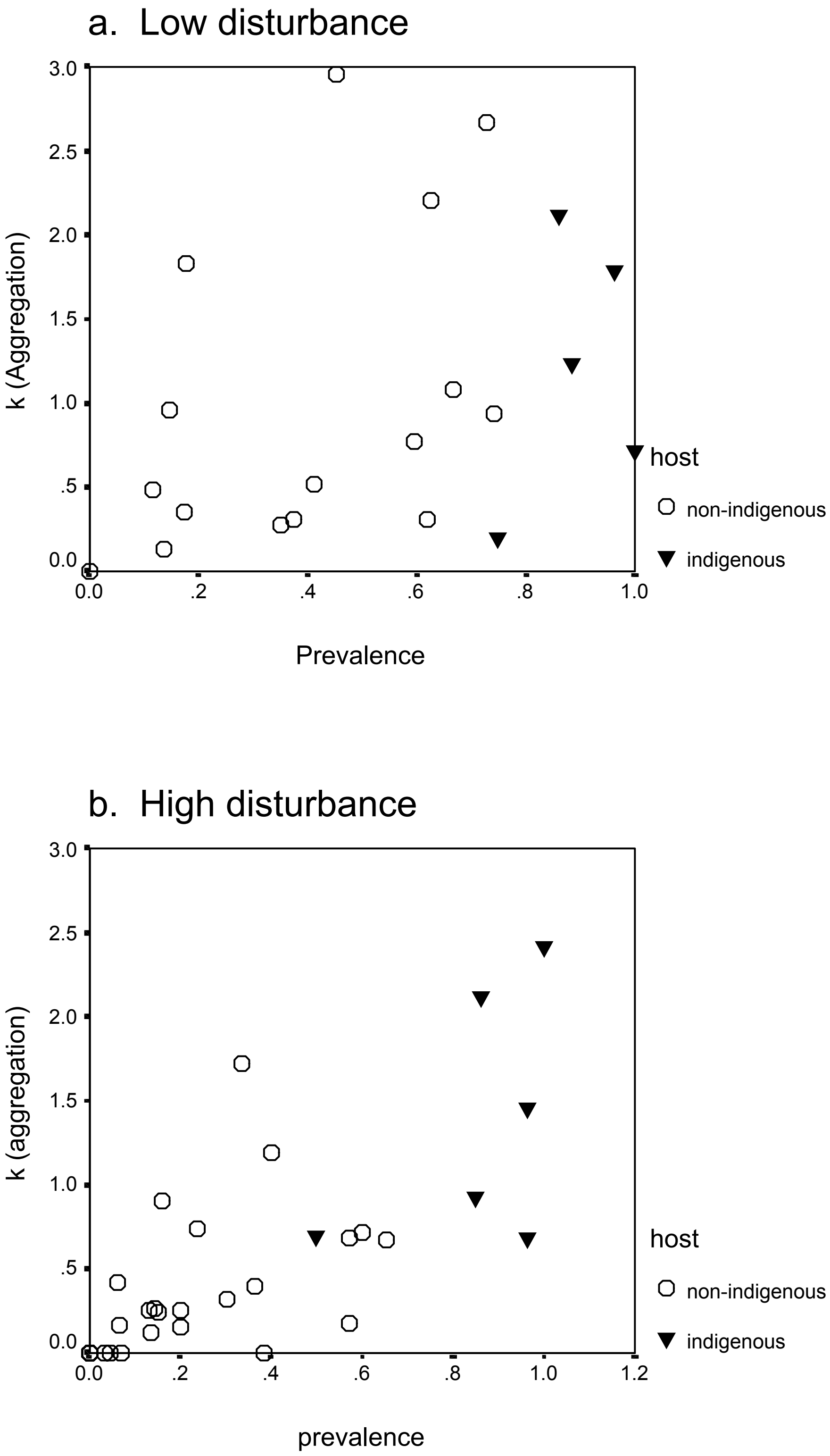
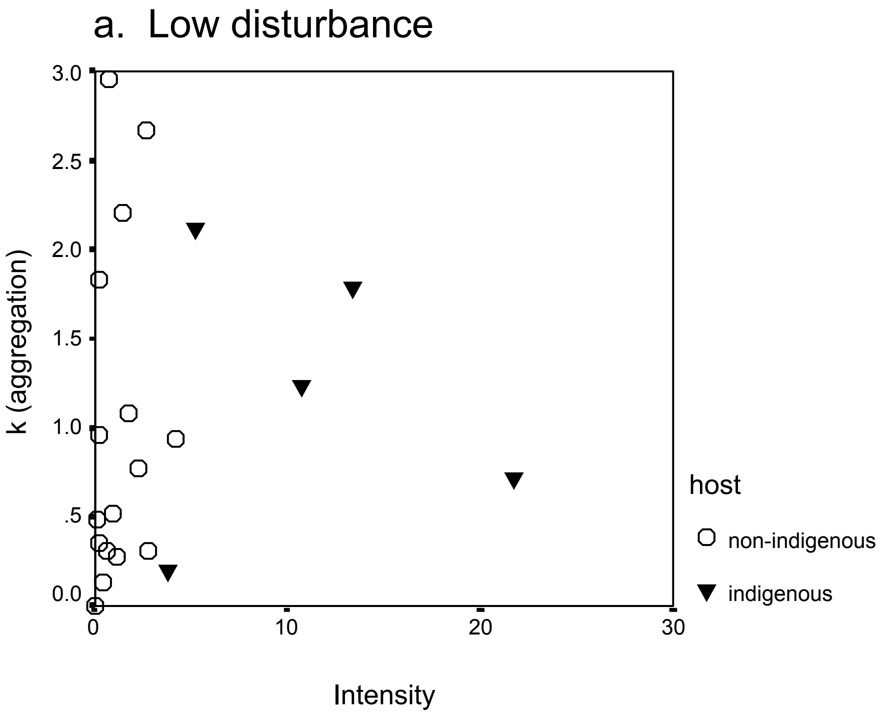


Figure 6.17 Change in aggregation (k) with prevalence for parasite communities of non-indigenous and indigenous freshwater fishes in northern Queensland: a, low disturbance habitats, b, high disturbance habitats



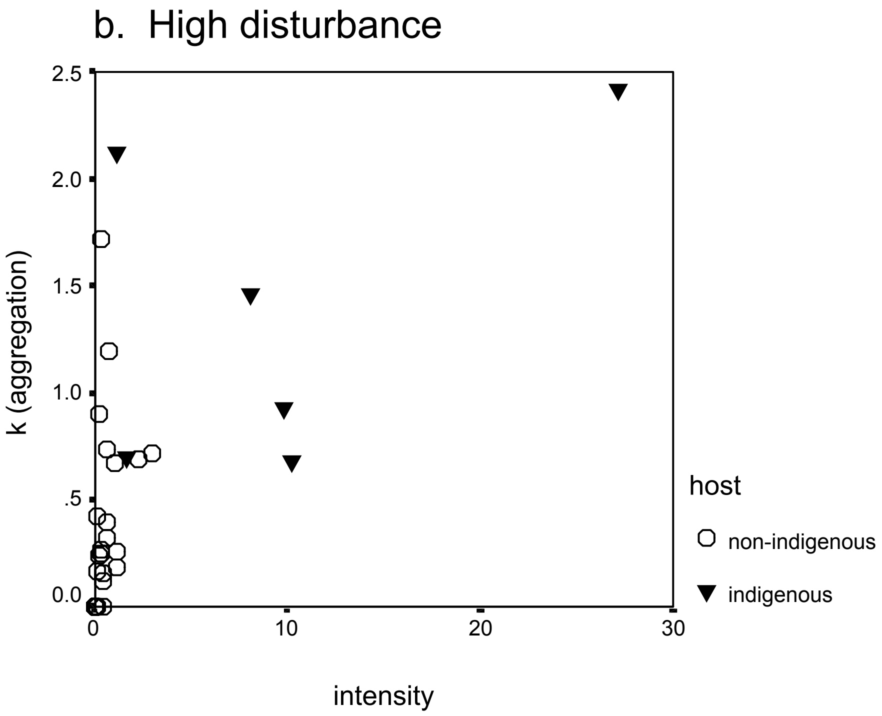


Figure 6.18 Change in parasite aggregation (k) with intensity for non-indigenous and indigenous freshwater fishes: a, low disturbance habitats, b, high disturbance habitats

Comparison of parasite species richness, aggregation, intensity, prevalence and frequency distributions of non-indigenous and indigenous host fish species between high and low disturbance habitats

For the three non-indigenous species, species richness was significantly less (1) in high disturbance habitats compared with low disturbance habitats (2.5) for only the Guppy (U = 11.000; p = 0.001). For the Mosquitofish, there was no significant difference in species richness between high disturbance habitats (1.0) and low disturbance habitats (1.0) (U = 27.00; p = 0.096). For the Mozambique mouthbrooder, there was no significant difference in median parasite species richness between low disturbance habitats (1.5) and high disturbance habitats (1.8) (U = 13.000; p = 0.792).

For the two poeciliids, the Guppy and Mosquitofish, parasite intensity, prevalence and *k* parameter values were significantly lower in high disturbance habitats (fewer and more lightly infected fish and more aggregated parasite dispersion), compared with low disturbance habitats. For the Mozambique mouthbrooder, there was no significant difference in intensity, prevalence or *k* parameter between high and low disturbance habitats (Table 6.10). Parasite frequency distributions per host fitted the negative binomial model for individual species in high and low disturbance habitats (Table 10). Frequency distributions for the species combined also fitted the negative binomial model for high disturbance habitats (χ^2 = 11.819; df = 8; p = 0.159) and low disturbance habitats (χ^2 = 12.662; df = 8; p = 0.124) (Figure 6.19).

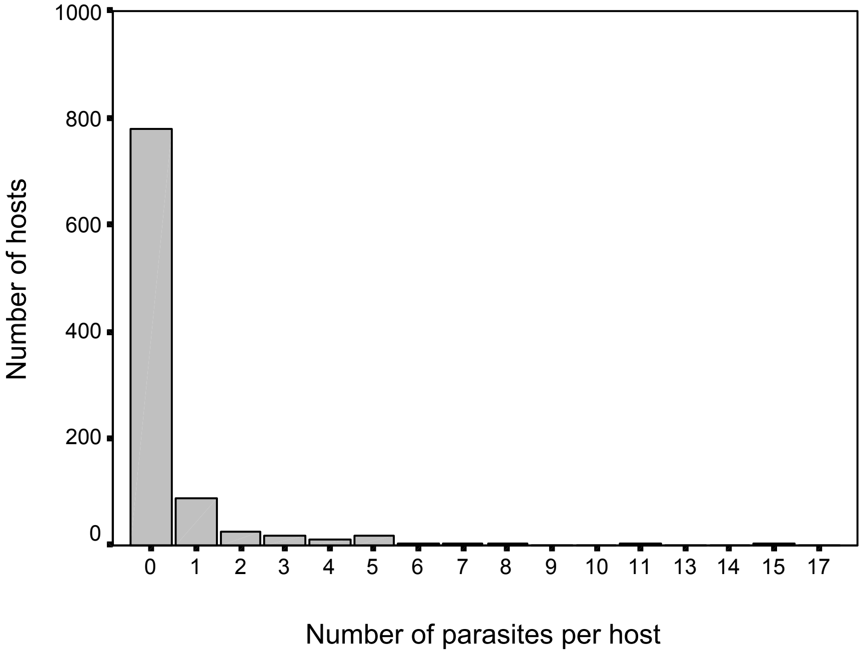
For the three indigenous fish species (Agassiz's glass perch, Fly-specked hardyhead and Empire gudgeon), there was no significant difference in parasite species richness between high and low disturbance habitats (U = 14.50; p = 0.926). There were significant differences in population parameters among these species (Table 6.10). Parasite frequency distributions per host for the three indigenous species combined did not fit the negative binomial model for high or low disturbance habitats (Figure 6.20). For the Empire gudgeon, there was no significant difference in intensity, prevalence and aggregation (*k*) parameter between high disturbance and low disturbance habitats. Parasite frequency distributions for the species also fitted the negative binomial model in both low disturbance and high disturbance habitats. Agassiz's glassperch had significantly more parasites per host (intensity) and less aggregated parasite dispersion in hosts (higher *k*) in high disturbance habitats compared with low disturbance habitats. The frequency distribution for the species differed significantly from the negative binomial model in both high and low disturbance habitats (Table 6.10).

Table 6.10 Statistical summary for comparison of parasite intensity, prevalence, aggregation (*k*) and goodness of fit to the negative binomial model for parasite communities of indigenous and non-indigenous freshwater fish species between high and low disturbance habitats in northern Queensland

Host	**Disturb**	**G. INTENSITY**			**H. PREVALENCE**			**I. AGGREGATION**			**NEG. BINOM.**		
		Inten.	U	p	Prev.	U	p	*k*	U	p	χ^2	df	p
MM	HIGH	0.66	10.00	0.357	0.452	10.0	0.357	0.7016	12.00	0.916	10.954	6	0.090
	LOW	0.21			0.172			0.4850			1.976	3	0.577
GAM	HIGH	0.04	9.50	0.013	0.04	7.00	0.009	0.0000	6.00	0.005	3.843	4	0.428
	LOW	1.02			0.50			0.9486			3.765	4	0.439
PR	HIGH	0.00	5.00	<0.001 *	0.00	7.00	<0.001*	0.000	11.00	0.002	0.113	3	0.990
	LOW	0.78			0.595			0.523			6.985	6	0.322
GP	HIGH	22.8	2.00	0.011	1.00	10.0	0.260	2.038	15.0	0.710	148.85	6	<0.001
	LOW	12			1.00			2.465			46.550	5	<0.001
HH	HIGH	9.35	3	0.067	0.97	6.0	0.257	1.391	6.0	0.257	17.937	8	0.020
	LOW	22.00			1.00			0.681			21.432	7	0.003
EG	HIGH	5.67	4.00	0.480	0.856	5.00	0.719	0.8109	5.00	0.857	10.064	7	0.185
	LOW	7.42			0.859			1.7912			7.206	6	0.302

Hosts: MM = Mozambique mouthbrooder; GAM = Mosquitofish; PR = Guppy; GP = Agassiz's glass perch; HH = Fly-specked hardyhead; EG = Empire gudgeon

a. Non-indigenous fish (high disturbance)



b. Non-indigenous fish (low disturbance)

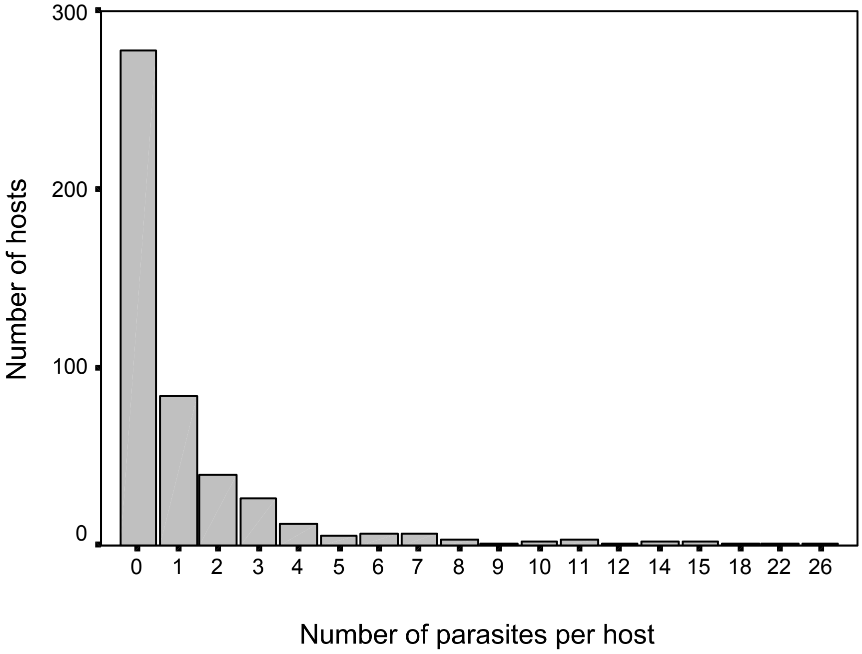
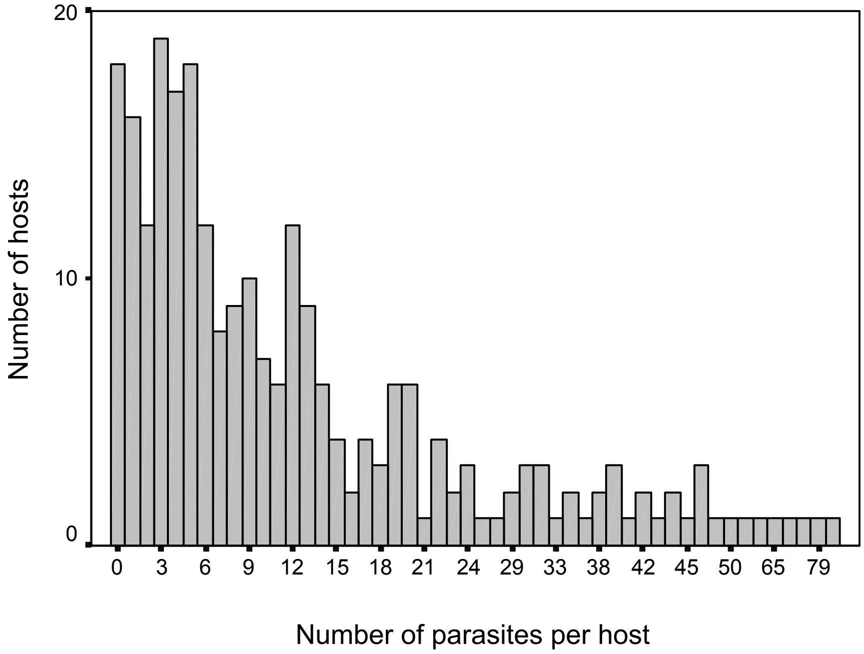


Figure 6.19 Parasite frequency distribution (all taxa) per host for non-indigenous freshwater fishes: a, high disturbance habitats, b, low disturbance habitats

a. Indigenous fish (high disturbance)



b. Indigenous fish (low disturbance)

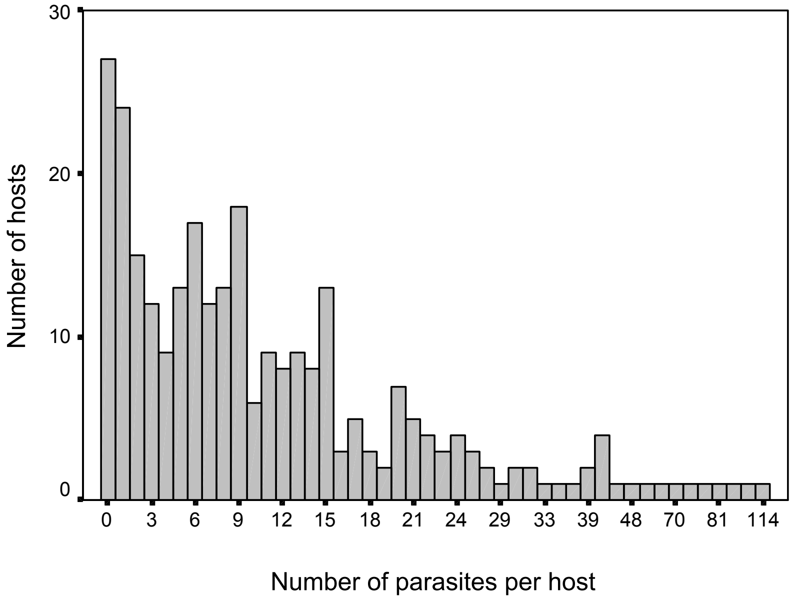


Figure 6.20 Parasite frequency distribution (all taxa) per host for indigenous freshwater fishes: a, high disturbance habitats, b, low disturbance habitats

In contrast, the Fly-specked hardyhead had significantly higher parasite intensity in low disturbance habitats compared with high disturbance habitats. The parasite frequency distribution for the species in high disturbance habitats fitted the negative binomial model and was more aggregated (lower k) compared with the frequency distribution for hosts in low disturbance habitats that differed significantly from the negative binomial model (Table 6.10).

6.4 Discussion

6.4.1 Parasite community structure of non-indigenous fishes in northern Queensland

Regional communities and infracommunities of parasites on non-indigenous fishes were markedly different from those of equivalent indigenous fishes in parasite richness, prevalence and intensity and patterns of aggregation on the host. Parasite communities of non-indigenous hosts were composed of low numbers (usually one) of specialist parasite species introduced with the host (core species with high prevalences), with infections of native generalist parasites (satellite species with low prevalences). Non-indigenous fishes in northern Queensland had significantly poorer infra- and regional parasite communities, with much lower interactivity and reduced importance of specialist core species than equivalent indigenous fishes. Since the probability that potential invaders harbour a significant proportion of exotic parasites will depend on the origin, size of subset population and effectiveness of quarantine procedures (see Evans and Lester 2001) these communities of non-indigenous fishes can be described as stochastically determined. These data are consistent with the theoretical models of Dogiel (1939) (in Petrushevski 1961), Dogiel (1948), Petrushevski 1961), Bauer (1991) and Kennedy and Bush (1994) for non-indigenous fishes following their introduction in new habitat, and are similar to results obtained by Dove (1999) for non-indigenous poeciliids in southeastern Queensland. These data, and the slow rate of parasite acquisition by non-indigenous fishes, differences in patterns of parasite aggregation and responses of parasite communities to habitat disturbance, also support the view expressed by Dove (1999) that non-indigenous fishes may have a competitive advantage over indigenous fishes by reduction in the adverse affects of parasites on host fitness.

Parasite richness recorded here for non-indigenous fishes in northern Queensland was an underestimate as microparasites and larger adult hosts were not included in the comparative analysis, the latter to minimise any size-related differences in parasite infection. Studies by Webb (1994) and observations in this study showed that adult Mozambique mouthbrooder had remarkably few parasites in freshwater habitats, and possibly reflect its largely phytophagous/detritivorous feeding habit. Indigenous fishes were sampled over a narrower geographical range of sites with fewer replicate samples compared with the non-indigenous fishes, yet still had significantly higher regional parasite richness, prevalences and parasite intensities compared with non-indigenous hosts. The non-indigenous host communities were dominated by monogeneans introduced with the host and indigenous ILC parasites, typically immature larval stages of low host specificity (e.g., *Clinostomum* metacercariae).

Kennedy and Bush (1994) described non-indigenous fishes, away from their "heartland" where they evolved, as "strangers in a strange land" without their co-evolved parasite fauna and encountering a taxonomically, if not ecologically new suite of parasites. This is an apt description, at least in the initial phase of introduction and establishment, where dissimilarity in geographic habitat and host habitat has a negative effect on parasite exchange, not only from indigenous to non-indigenous fishes but also *vice versa.* In this study, although parasite origin was, in many cases, inferred and based on *a priori* decisions, the direction of transmission was almost entirely from indigenous to introduced host. At least 15 species of indigenous parasites compared with five non-indigenous parasites were found on non-indigenous host fish. This proportion is similar to that found by Dove (1999) in southeastern Queensland. These parasites, however, represented only a very small proportion of the total number recorded for indigenous fish hosts during the study. Also, the majority of the indigenous parasites were found in low prevalences and intensities, or as single occurrences on non-indigenous hosts, the non-indigenous monogeneans being the dominant core species on non-indigenous hosts.

Dove (1999) found two exotic parasites, *Bothriocephalus acheilognathi*, and the ciliate, *Trichodina heterodentata,* making a much larger contribution to the structure of parasite communities in indigenous fishes than did indigenous parasite species make to parasite communities of exotic fishes. In the present study, specimens of *B. acheilognathi* were only found in non-indigenous fish hosts, the Guppy and Mozambique mouthbrooder. While indigenous fish at the same location were not sampled, they were most probably infected, although it is unlikely their contribution would have made a major difference to the regional parasite community structure if currently restricted to a single creek. The majority of microparasites, including protists, such as trichodinids, were not included in this study, so the potential contribution of these taxa was not determined.

The original source in northern Queensland of *B. acheilognathi* is not known, although it was first imported into Australia with European carp and is known to have been transmitted to indigenous hosts (Dove *et al.* 1997). No Carp, or other cyprinids (excluding the unidentified specimen from the Ross River) have been reported from open waters in northern Queensland, although populations of Koi carp have been reported from ponds in the Cairns region (D.J. Russell, Northern Fisheries Centre, Cairns, *pers. comm.*). Infected Guppies and Mozambique mouthbrooder may be escapees from such ponds, or there may be an undetected Carp population in Wright Creek. Dove (1999) noted that infection of indigenous fish by this parasite is not possible without the presence of an exotic host as a reservoir of infection. Damage to indigenous fish can therefore be ascribed directly to the presence of the exotic fish, whether the reservoir is Carp or other non-indigenous fishes is unknown. Evans and Lester (2001) found that imported shipments of Guppies and Platies into. While the impact on either non-indigenous or indigenous fish populations in Wright Australia were infected with the parasite which suggests that non-cyprinid hosts can act as reservoirs Creek is not known, *B. acheilognathi* is known to cause significant pathology in newly acquired hosts while reservoir hosts remain relatively unaffected (Dove 1999). The latter non-indigenous hosts may therefore be at a competitive advantage due to reservoir dependent parasite suppression of indigenous.

At least four other non-indigenous parasite species were found on non-indigenous host fishes. All of these parasites were monogeneans, three of which were previously recorded on poeciliids by Dove (1999) and one from the cichlid, Mozambique mouthbrooder. The localised distributions of these non-indigenous parasites reflect both historical and stochastic processes: the source of host fish and quarantine protocols applied, the pattern of introductions and subsequent degree of isolation of these populations. For example, the monogenean, *Gyrodactylus bullatarudis* was found on Guppies only in the Cairns region, while the monogenean, *Salsuginus heteroclitis*, was only found on Mosquitofish in the Townsville region, although both host species were widely distributed in northern Queensland, particularly in the 1940s, for mosquito control. The monogenean, *Cichlidogyrus tilapiae*, was only found on Mozambique mouthbrooder in the Townsville region and was absent from this species in the Cairns region and possibly reflecting differences in the origins of the founding stocks. While the Townsville population is pure *Oreochromis mossambicus*, and probably originated from wild stocks brought to Australia direct from Africa, the Cairns population is a hybrid (the gold Zanzibar strain: *O. mossambicus x O. honorum*) produced for aquaculture and probably originated from Singapore (Lear 1987; Blühdorn *et al.* 1990).

The non-indigenous parasite fauna was dominated by specialist core species (monogeneans) rather than generalist species, as Dove (1999) found for poeciliids in southeastern Queensland. Dove noted that no poeciliids had imported any parasite species that infect native fishes. However, Evans and Lester (2001) found remarkably high prevalences of at least 10 parasite species in shipments of ornamental fish imported into Brisbane, even after a period of quarantine. These parasites included specialist and generalist ILC parasites known to cause serious pathology, notably *B. acheilognathi* and the nematode, *Camallanus cotti,* already transmitted to indigenous Australian fishes as they are non-specific for their intermediate hosts (copepods). The composition of regional faunas of non-indigenous parasites may simply reflect historical patterns of importation rather than the lack of, or low probability of, importing specific types of parasites. Clearly, ILC parasites with highly specific intermediate hosts, not found on non-indigenous fishes in the present study, require both the definitive and intermediate hosts to be imported and to establish in the same location before transmission to indigenous hosts is likely.

6.4.2 Aggregation of parasites on indigenous and non-indigenous fishes

Virtually all parasite taxa examined for both indigenous and non-indigenous hosts were aggregated and fitted the negative binomial model. However, the frequency distribution patterns for total parasite loadings showed differences between indigenous and non-indigenous hosts. All non-indigenous hosts had aggregated parasite distributions that fitted the negative binomial model, while only half the indigenous fish species had aggregated parasite distributions that fitted that model; the other species had multi-modal parasite distributions.

These differences in aggregation reflect the lower parasite richness, prevalence and intensity of the parasite communities of non-indigenous fishes compared with indigenous fishes. Host pathology reflects a complex interaction of various intrinsic factors, such as the type of parasites present, host or parasite size and location in the host, and extrinsic factors such as increased susceptibility to infection due to external stresses, including water pollution (Lafferty and Kuris 1999).

Differences in host condition or mortality are very difficult to determine in natural host populations and were not directly assessed in the present study, nor were the relative effects on competitive outcomes of interactions between indigenous and non-indigenous fish hosts. However, if parasites have, by definition, a negative impact on the host and these impacts are additive (Holmes 1979, 1982; Sorci and Clobert 1995) and condition of the host is negatively correlated with parasite load (Lester 1977; Herbert and Isham 2000; Karlsbakk 2001), then fishes with greater species richness, higher prevalences and higher intensities (i.e., indigenous fishes) are more likely to be exposed to adverse density-dependent effects of parasites (Guiyan 1996). Barnard (1990) also suggested that aggregation might have an inhibitory effect on the rate of transmission of parasites to other hosts if the parasites only adapt to a small proportion of host genotypes in each generation. Non-indigenous fish populations, therefore, by having highly aggregated parasite communities – that is, with many fish uninfected or with very few parasite present - are advantaged in terms of their fitness with a relatively smaller proportion of their population exposed to higher levels of infection compared with at least some of the indigenous fish species examined in the study.

6.4.3 Effects of habitat disturbance on parasite community structure of indigenous and non-indigenous fishes

In high disturbance habitats, both indigenous and non-indigenous hosts had similarly aggregated parasite frequency distributions, which reflected a decrease in parasite species richness, intensity and prevalence. In low disturbance habitats, non-indigenous fishes had significantly lower species richness, intensity and prevalence and more aggregated parasite distribution compared with indigenous fishes. For the latter, the frequency distribution did not fit the negative binomial model, rather a multi-modal distribution due to the greater species richness, intensity and prevalence of parasites in indigenous hosts.

There were interesting differences in patterns of parasite aggregation among non-indigenous and indigenous host fish species, which may reflect individual host responses to changed conditions or changes in availability of parasite faunas, for example due to the loss or greater presence of intermediate hosts of ILC parasites. The non-indigenous Guppy and Platy showed strongly aggregated parasite distributions and significantly lower parasite richness, intensity and prevalence in high disturbance habitats compared with low disturbance habitats, while the Mozambique mouthbrooder showed no significant difference in these parameters with habitat disturbance. This may reflect differences in diet between the more herbivorous cichlid and the more carnivorous poeciliids and therefore exposure to particular groups of parasites. It may also be due to loss of particular groups of parasites in high disturbance sites more likely to infect the poeciliids than the cichlid.

The parasite distributions of the two indigenous hosts examined (Agassiz’s glass perch and Fly-specked hardyhead) were very different and may reflect opposite responses of either host or parasites to habitat disturbance. Agassiz’s glassperch appeared to be more susceptible to infection with less aggregated parasite distributions, due to higher parasite intensities and prevalence and higher species richness in High disturbance habitats. The Fly-specked hardyhead showed a similar pattern, but for low disturbance habitats; in high disturbance habitats this species may have lost some of its parasite fauna due, for example, to loss of intermediate hosts. The Empire gudgeon showed no difference in intensity, prevalence or species richness and with frequency distributions highly aggregated in both high and low disturbance habitats. This species has a predominantly carnivorous diet similar to the Eastern Queensland rainbowfish and Agassiz’s glass perch, feeding on microcrustaceans and insect larvae (Allen 1989, 1996; Herbert and Peeters 1995). While it may be exposed to similar parasites to rainbowfish and glassperch, it is one of the hardiest indigenous freshwater fishes and is often found in dense vegetation (Herbert and Peeters 1995; Growns *et al*. 2003) and in disturbed or unstable habitats, such as overgrown lagoons or shallow pools (see Chapter 4) that may be too extreme for some parasites or their intermediate hosts, or reflect a greater resistance of the species (i.e., greater stress thresholds) to infection.

Prior to, and upon arrival, non-indigenous invaders are subject to similar processes that result in parasite reduction and subsequently much poorer parasite communities compared with their homeland faunas and also with resident indigenous fishes. Host-parasite interactions are clearly complex and non-indigenous and indigenous fish communities are not homogenous entities; species differ, irrespective of origin, with respect to their ecological niches, particularly trophic relationships, differing tolerances to environmental gradients and their capacity to tolerate or to exploit changed conditions.

In this study, habitats were defined as fitting into one of two categories: high and low disturbance, based on a qualitative assessment of site condition. However, further categorisation of disturbance type is warranted to elucidate changes in host–parasite interactions and subsequent changes in parasite communities of both indigenous and non-indigenous fishes. For example, disturbance may involve added resources, such as eutrophication resulting in proliferation of aquatic vegetation; or it may involve resource removal, such as pollution resulting in a loss of habitat or faunal diversity. The former can lead to a proliferation of primary consumers, such as molluscs and crustaceans that are intermediate hosts for a range of parasites, notably digeneans, and therefore increased likelihood of infection by these parasites (Zander *et al*. 2000). Some changes in water quality (e.g., acidification), may adversely affect parasites more than hosts and lead to a reduction in the occurrence of parasites in the polluted waters (Marcogliese and Cone 1996, 1997).

6.4.4 Parasite acquisition by non-indigenous fishes in northern Queensland

While there were limitations in the data due to small sample sizes, uncertainty regarding the exact timing of some introductions and lack of comprehensive records of regional parasite communities for all non-indigenous fishes, there was a strong, positive correlation between estimated residency time and the number of indigenous parasites acquired by non-indigenous fish hosts in northern Queensland fresh waters. Residence time was shown to be an important if not exclusive factor in determining parasite acquisition. Encounter rates and acquisition of parasites by hosts will be influenced by factors such as diet, host size, geographic distribution and host specificity of the parasite (Vickery and Poulin 1998), leading to differences between ecologically non-equivalent species, such as a specialised herbivore and more generalised omnivore or carnivore (Poulin 1995). Latitudinal differences in parasite acquisition rate also are important: the omnivorous Goldfish has a residency time in southern Australian waters of about 127 years while the Mozambique mouthbrooder, another omnivore, has a residency time in tropical northern Queensland of only about 25 years, but juveniles of the latter species have acquired about twice the number of parasites of the similar-sized Goldfish.

A linear model can describe the early phase of acquisition of new parasites by non-indigenous fishes in both tropical and temperate regions of Australia. Guegan and Kennedy (1993) found that residency time was an important determinant of helminth species richness of non-indigenous fishes in Great Britain and also suggested that for unsaturated hosts with few parasites during the early stages of establishment, infracommunity and regional richness are linearly related to residency time, although the relationship, on a larger time scale, is probably curvilinear. While there were limitations in the data due to small sample sizes, the underestimates for Brown trout residency times support this observation: that the larger time-scale relationship is probably non-linear and is better described by a power function, such as the Weibull growth model (see Figure 6.11) or the Verhulst-Pearl logistic model (Odum 1971). The latter model incorporates an asymptote that might represent parasite saturation or carrying capacity of the host. According to Esch *et al*. 1977, parasites are located along an r-k continuum and population growth curves of individual parasites within an assemblage may approximate to either of the classic growth forms: the sigmoid form (k- strategist) or exponential form (r-strategist). On a geological time scale, changes in parasite community may be analogous, particularly as host saturation approaches and parasite community interactivity increase (i.e., density-dependent factors have greater effect) and slow the rate of parasite acquisition. While saturation may have occurred well before present, subsequent changes, where some parasites are lost and new ones gained, are probably rare events and stochastic in nature. There might therefore be relatively little change in parasite community richness over long periods of time (Vickery and Poulin 1998), and these changes occur at much slower rates than during the earlier, unsaturated phase.

The results suggest that parasite acquisition is a very slow process and should be measured in centuries or millennia rather than decades. Latitudinal differences may reflect the greater diversity of both host and parasite communities in tropical northern Queensland in comparison with the communities of more temperate regions examined, and therefore a greater encounter rate between hosts and parasites in the tropics. Non-indigenous hosts are temporarily unsaturated due to stochastic reduction of their coevolved parasite fauna immediately before and after arrival in a new habitat. The relatively slow rates of parasite acquisition by non-indigenous fishes might therefore moderate the potentially adverse and additive impacts of parasites on host fitness and confer non-indigenous fishes with a competitive advantage over resident fishes.

6.4.5 Conclusion

Non-indigenous fishes in northern Queensland have depauperate, stochastically determined parasite communities dominated by DLC parasites (monogeneans) that were introduced with the host fishes. Non-indigenous fishes had significantly fewer parasites than indigenous fishes even though the observed direction of transfer of parasites was from indigenous to non-indigenous hosts. Habitat disturbance has a key role to play in the invasion process by altering ecosystem dynamics that might provide an "opening" for opportunistic, healthy invaders freed from their parasite burdens and capable of taking advantage of the changed conditions. To reflect this importance, and the increasing frequency and extent of disturbance to freshwater systems, the influence of habitat disturbance in the invasion process was included in Moyle and Light's model (Figure 6.1). While there are differences between host species irrespective of origin or habitat disturbance, there are features of host-parasite interactions, which characterise the invasion of non-indigenous fishes. If parasites have a discernible and additive effect on host fitness, indigenous fishes may be at a competitive disadvantage in the presence of non-indigenous fishes due to:

- reduction in richness of parasite communities of non-indigenous fishes immediately before and after arrival;
- infection of naïve indigenous fishes by non-indigenous parasites introduced with their hosts and representing a serious pathogenic risk (e.g., *B. acheilognathi*);
- more aggregated parasite frequency distributions in non-indigenous fishes reducing density-dependant regulation of hosts by parasites due to lower parasite intensity and prevalence, especially in disturbed habitats; and
- the very slow rate of acquisition of new parasites by "unsaturated" non-indigenous fishes.

As Dove (1999) succinctly stated, "regardless of the parasite status of exotic fishes, native fishes will nearly always be at a disadvantage" (p. 219). In the context of predation and resource competition the outcome is almost certainly that at least some indigenous species will lose out in the presence of exotic fishes.

CHAPTER SEVEN

GENERAL CONCLUSIONS

"So long and thanks for all the fish"

Douglas Adams, 1979
Hitchhiker's Guide to the Galaxy

7.1 Distribution of non-indigenous fishes in northern Queensland

The introduction of non-indigenous fishes into open waters in northern Queensland waters is continuing with 17 species now reported from northern Queensland. At least 10 of these species have established breeding populations and the majority has been present for more than 10 years. These fish represent at least fifty percent of all non-indigenous fishes reported in Australian waters and the Ross River catchment now has the highest number of non-indigenous species reported of any waterway in Australia. These species probably represent only a small fraction of all introductions, with many not able to establish feral populations. However, the number of non-indigenous species introduced into Australian waters in the past 30 years is more than triple the number introduced in the preceding 110 years (see Appendix B, Figure B1) and the rate of increase in the latter part of the 20th century is almost exponential. It is clear that if the trend continues, then in the coming decades, non-indigenous fishes may become the dominant component of our fish communities. Localised dispersal is occurring with some species, although large increases in range are due to human translocation. There is an urgent need to prevent further introductions of these fishes and their subsequent spread that add, either directly or indirectly, to the ecological pressure on many indigenous species already under threat through habitat loss or degradation.

There was a positive association between the habitat matching, propagule pressure indices and previous history of introductions elsewhere and success in establishment in northern Queensland. Most fishes used in the aquarium trade are tropical species, so climate conditions in northern Queensland are "just like home" for them. Many fish have been selected as ornamental species because of their hardiness and often omnivorous habit that are, for the aquarium trade, positive economic attributes. Increased frequency of introductions (as an index of propagule pressure) will increase the probability that favourable conditions are encountered by an invader. This "pressure" might also reflect the "usefulness" of the species to humans – for example, the Mozambique mouthbrooder is one of the most widely distributed fish species globally and has been used for research, for commercial fisheries and aquaculture, as a bait fish, as an ornamental species and as a biological control agent (mosquitoes and aquatic weeds) (Hofstede and Botke 1950; Hauser *et al.* 1976; Moyle 1976; Philippart and Ruwet 1982; Welcomme 1988, 1992).

Computer models based on climatic matching, such as BIOCLIM/ANUCLIM and CLIMEX have been used to predict the distribution of introduced terrestrial flora and fauna, although these models have not yet been applied to non-indigenous fishes. While such models can make better predictions compared with the relatively simple indices used in this study, their predictive power is still less than needed for practical management application. While absolute prediction of establishment of any "invader" is unlikely, from a management perspective, the refinement of models, e.g., the relatively user-friendly, taxonomic filter model of Arthington *et al.* (1999), to improve predictive power will be of significant benefit. There is limited ecological information available not only for species imported into Australia and maintained in aquaria, but for many of the species with feral populations in Australian waters and elsewhere. Such information, particularly on physiological tolerances, is of environmental and commercial importance when import of non-indigenous fishes requires risk assessment. Such detailed and systematic ecological information will allow more consistent, rational decision making to minimise importation of a potentially invasive or pest species while maintaining a viable industry and popular recreational activity.

7.2 Invasion of non-indigenous fishes in northern Queensland and habitat disturbance

Virtually all of the non-indigenous species reported in northern Queensland occupy waterways in both agricultural and urban environments that have suffered disturbances to hydrological conditions, including reduction in water quality through pollution and altered water flows, by changes to stream morphology and construction of barrages and dams. There have also been subsequent changes to in-stream and adjacent riparian vegetation, with the loss of indigenous vegetation and its replacement by non-indigenous plants, or proliferation of both in nutrient enriched conditions. These conditions have also resulted in declines of local populations of indigenous fishes.

The model by Moyle and Light (1996a) (see Figure 1.1) provides a sound framework for understanding invasion processes. Moyle and Light argued that outcomes of invasions are dependent on "favourable environmental conditions at the time of invasion and only secondarily on invading interactions between invading and resident fishes" (p. 152). Clearly, an invader will not persist if it cannot tolerate, or is unable to adapt to, existing conditions, such as flow regime, temperature or pH ranges. Moyle and Light recognised that the complex interaction of abiotic, biotic and demographic factors that provide the “resistance” to potential invaders can be compromised by anthropogenic habitat disturbance. For example, dispersal routes for invaders can be facilitated by construction of canals, irrigation channels – and even by fish passages around barriers to dispersal such as dams and weirs; these barriers can modify stream flows and provide conditions favouring those fish better adapted to lacustrine conditions and changes in fish assemblages might advantage invaders if resident predators or competitors are lost.

Laboratory and field observations in this study support the hypothesis that such habitat disturbance plays a significant role in the establishment and persistence of non-indigenous fishes. The results showed that indigenous predators select prey on the basis of morphology and behaviour rather than place of origin: irrespective of foraging mode, they are optimal foragers and, given the opportunity, will consume non-indigenous prey such as Mosquitofish and the Mozambique mouthbrooder. These species are advantaged where changed non-natural conditions not only provide a familiar habitat (e.g., dense vegetation, including exotic plants), but also create a physical and hypoxic barrier that reduces potential interactions with native biota, including predators, parasites and competitors.

These invaders may be doubly advantaged upon arrival by “losing” most of their co-evolved parasite fauna. This study supports the model provided by Dove (1999) to explain parasite richness processes as non-indigenous fish hosts enter new habitat, and how indigenous resident fishes might be disadvantaged. Significantly reduced parasite species richness, increased aggregation of parasites on hosts with much lower parasite loads and higher numbers of uninfected fish were observed for non-indigenous fishes compared with indigenous fishes. Such changes can provide non-indigenous fishes with a relative advantage by minimising the density-dependent regulation of hosts by parasites and thereby enhancing their fitness. The results also showed that the acquisition process is very slow and provides an extended period when hosts are unsaturated. The rate of host fitness reduction by parasites (or host “adjustment”) is therefore gradual and reflects a co-evolutionary time scale of the host-parasite system that should be measured in centuries in the tropics (even millenia in temperate regions) rather than in decades.

The results also showed that habitat disturbance can significantly alter the parasite communities of non-indigenous fishes, with even greater aggregation, reduced species richness and intensity and increased prevalences compared with indigenous fishes. While there were clear overall differences for the two groups of fishes, the results indicate that individual host species (and parasites) undoubtedly respond differently to disturbance. While the study did not differentiate different types of disturbance, rather the level of disturbance (low or high), the effect on host-parasite interaction will be a combination of both factors. For example, proliferation of vegetation may favour intermediate hosts of many ILC parasites, such as molluscs or crustaceans, that might lead to an increase in encounter rates with these parasites. On the other hand, high levels of chemical pollution may have adverse affects on either ILC or DLC parasites or vectors of the former, or increase host susceptibility to infection. While the poeciliids examined (Guppy and Mosquitofish) benefited by reduced parasite richness and intensity, increased prevalence and aggregation in high disturbance habitats, there was no significant difference observed for the Mozambique mouthbrooder. This may reflect trophic differences between the poecilliids (omnivores but predominantly carnivores) compared with the Mozambique mouthbrooder (omnivore but predominantly herbivorous/detritivorous) and therefore differences in parasite exposure. Further studies are needed to clearly differentiate between disturbance type and level of disturbance and its effect on parasite burdens of host fishes. The results, however, do suggest that habitat disturbance, besides providing refuge from predators and competitors, can benefit non-indigenous fishes by further reducing their parasite loads and thus enhance their fitness.

Non-indigenous fishes also may disadvantage indigenous fish populations by introduction of pathogens, especially viruses and bacteria but also a number of metazoan parasites, which are capable of causing significant pathology. Dove (1998) aptly described such pathogens as the “silent tragedy” that continue still to enter undetected (see Evans and Lester 2001), and can be responsible for large-scale mortalities to indigenous hosts. Of particular concern was the discovery during the study of the exotic cestode, *Bothriocephalus acheilognathi,* in samples of Mozambique mouthbrooder and Guppies collected in the Cairns region. This cestode, brought into Australia with European carp, is known to cause serious pathology with mass mortality in both cultured and wild fish populations in several countries (Boomker *et al*. 1980; Weirowski 1984; Dove *et al*. 1997; Heckman 2000). The current distribution, origin and possible impacts of this parasite on fish populations in northern Queensland warrants further investigation.

7.3 Management of non-indigenous fishes in Queensland fresh waters

The results suggest that aquatic habitat rehabilitation in urban and agricultural waterways in northern Queensland may be an effective tool in the longer-term management of non-indigenous fish species. Removal of dense vegetation (and amelioration of the conditions responsible for proliferation), and therefore the physical and hypoxic refuge it provides, will make non-indigenous species more vulnerable to predation and also expose them to increased competition from resident fish. While not necessarily recreating conditions prior to European settlement, such restoration can facilitate natural hydrological processes and achieve what Higgs (1996) described as "functional success" that favours the re-establishment of healthy indigenous fish communities necessary to provide the biotic resistance to invasion by non-indigenous species.

In the Murray-Darling Basin, researchers have now recognised river conditions that have dramatically reduced indigenous fish numbers also favour many introduced species such as European carp (Harris 1997; Barrett 2003). The Native Fish Strategy of the Murray Darling Basin Commission (MDBC) is a long term, integrated program of restoration emphasising catchment management, flow allocation, pollution abatement, habitat reconstruction and restoration of connectivity disrupted by barriers to help to reduce Carp populations and benefit indigenous species. Incorporated within the National Management Strategy for Carp Control 2000-2005, developed by the MDBC, are studies to restore fish habitat. These include resnagging to provide refuge and spawning sites (Lloyd and Walker 1988), as well as rehabilitating the wet/dry cycles of floodplain wetlands critical to the reproductive cycles of many indigenous fishes within the system (Geddes and Puckridge 1988). Rehabilitation includes restoration of riparian habitat, the absence or alteration of which can have a range of adverse impacts on indigenous fishes, such as disruption of reproductive cycles and metabolic processes, increased mortality rates, lowered disease resistance, and can facilitate invasion of non-indigenous species especially where exotic pasture grasses have proliferated (Pusey and Arthington 2003) (see Chapter 4). These researchers noted that large-scale terrestrial restoration projects are underway in northern Australia, but rarely are intended to rehabilitate stream ecosystems other than "improving water quality and, even more rarely, are stream fish, lacking the charismatic profile of vertebrate taxa such as cassowaries or tree kangaroos, the target taxa" (p.10). In view of the important linkages between fish and riparian vegetation, consideration of stream ecosystems in such restoration projects are potentially of great importance for conservation of freshwater biodiversity of northern Australia.

Attempts at biological control of non-indigenous organisms have largely focussed on those that, besides indigenous organisms, adversely affect agricultural or forestry production, with most success achieved with control of introduced weeds and invertebrates pests (Kogan 1986; Mengech *et al*. 1995). Fish have largely been used as biological control agents in public health programs to target vectors of pathogens causing human diseases such as malaria, schistosomiasis and filariasis (Slootweg *et al* 1994; Stauffer *et al*. 1997; Mishra and Saxena 2000) or used in biomanipulation programs to improve water quality by reduction, for example, of excessive macrophyte growth or phytoplankton blooms (Starling 1993; Liebeman 1996). The control of "pest" or unwanted fish in more open waters has largely been limited due, in part, to the lesser economic importance of freshwater fisheries resources especially in industrialised nations, but also, ironically, to the necessity of developing these non-indigenous resources in nations where protein production is critically important for human survival. This has resulted in certain fish, such as the oreochromid cichlids, being regarded as a noxious species in Australia under the Queensland Fisheries Regulation 1995, but of great commercial benefit in others, such as Sri Lanka (de Zylva 1999). Piscivorous fishes have mainly been stocked in aquaculture ponds, with some success due largely to the nature of the habitat (small water volume and absence of refuges) (Ghosh *et al.* 1989). Control by piscivores in larger waterbodies has had variable success due to a number of factors including stocking density, target specificity of the predator and degree of physical structure of the habitat (Grimm and Backx 1990; Bolding *et al* 1998; Mundahl *et al*. 1998)

Recent advances in gene research applied to fish by Australian CSIRO Marine Research scientists, however, show promise for "target specific" control by production of monosex (male) populations without the problems associated with use of introduced predators in the context of a more traditional pest control program. This "daughterless fish" technology is currently under development for eventual use within the Murray Darling Basin against Carp and Mosquitofish (Bax and Thresher 2003). The technology appears to have several advantages, such as its species specificity, its inheritability, it not involving insertion of foreign genetic material or genetically modified vectors (e.g., viruses), and it not eliciting an immune response. However, there are still ethical concerns about the use of such organisms and also the ecological risks to populations of the target species in their indigenous range should such manipulated agents be "exported". Furthermore, CSIRO researchers acknowledge that the technology, assuming it passes experimental and ethical tests, will be slow in effect, requiring decades rather than years to show significant results, and will not necessarily provide a guarantee of eradication.

There is still no "silver bullet", at least in the foreseeable future, that will provide an overnight solution to the problem of introduced organisms. In view of the environmental risks and ethical concerns, such technologies may also require extensive risk assessment and their application may be ultimately unacceptable. In contrast to biological control methods, habitat rehabilitation, although expensive and slow in effect, does not involve introduction of control organisms and arguably is a substantially smaller environmental risk – habitat rehabilitation can be a cornerstone of integrated catchment management involving a wide range of strategies, where natural populations of indigenous predators and competitors are then able to provide a dynamic, self regulating biotic environment that can reduce existing indigenous fish populations and inhibit establishment of new invaders.

Government in Queensland has recognised, in response to community concern, the significant problems presented by introduced, non-indigenous fishes and the future consequences of inaction. The Queensland Fisheries Service (QFS) initiated a comprehensive operational strategy for management of exotic pest fishes for Queensland fresh waters that complements the MDBC's Carp Control strategy (Mackenzie *et al.* 2000; Mackenzie 2003). The QFS strategy includes current reassessment of noxious species lists that are based on more consistent criteria to further restrict import of species potentially harmful to native aquatic communities (A. Burton, QFS, Brisbane, *pers. comm.*). This strategy is based on the principles of integrated pest management and, along with more rigorous import procedures and risk assessment for non-indigenous fishes being developed by AQIS (see Kahn *et al*. 1999; Arthington *et al.* 1999), provides a realistic and adaptive framework for management of non-indigenous fishes, while recognising that there is still a basic lack of ecological information on invaders and the invasion process. The strategy places particular emphasis on public education and community participation as essential to success. While the immediate focus is on direct control or eradication of exotic pest fishes, QFS has also recognised the need to incorporate such a strategy in a broader management plan which includes restoration of aquatic habitat and indigenous fish populations similar to that for the Murray Darling basin, but applied to waterways throughout Queensland (A. Burton, QFS, Brisbane *pers. comm.*).

The management of non-indigenous fishes in Australian waterways is a task that will require long term commitment of significant human and financial resources, and require the cooperative effort by all stakeholders, including resource managers, educators, hobbyists, members of the aquarium trade and researchers. To provide an informed and rational public debate and appropriate management options, such an enterprise needs to be underpinned by scientific research and evaluation, commendably a principle goal of the QFS strategy. The results of this study have added to current knowledge of the invasion process of non-indigenous fishes and advanced the need for management to be ecologically informed. While we seek to reduce invasion to a simple model and improve predictability, it is clearly a complex process: it may be inherently stochastic making prediction of outcomes very difficult, or our current understanding is simply limited by lack of information. There is, therefore, much work still to be done on the ecology of non-indigenous fishes to assist in the further development of effective risk assessment and management strategies.

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APPENDIX A

Table A1 List of fish species recorded from the freshwater reaches of the Ross River catchment above Aplin Weir (Milward and Webb 1990; Webb 1994 and unpubl. data prior to this study) (*tr = translocated species)

Indigenous species

Ambassis agassizi (Steindachner, 1867)
*Amniataba percoides (*Günther, 1864*)*
Anguilla reinhardti (Steindachner, 1867)
Craterocephalus stercusmuscarum stercusmuscarum (Günther, 1867)
Glossamia aprion gillii (Richardson, 1842)
Hephaestus fuliginosus (Macleay, 1883) *tr
Hypseleotris compressa (Krefft, 1864)
Hypseleotris galii (Ogilby, 1898)
Lates calcarifer (Bloch, 1790)* tr
Leiopotherapon unicolor (Günther, 1859)
Megalops cyprinoides (Broussonet, 1772) *tr
Melanotaenia splendida splendida (Peters, 1866)
Mogurnda adspersa (Castelnau, 1878)
Nematolosa erebi (Günther, 1868)
Neosilurus ater (Perugia, 1894)
Neosilurus hyrtlii Steindachner, 1867
Oxyeleotris lineolatus (Steindachner, 1867) *tr
Rediogobius bikolanus (Herre, 1927)
Awaous acritosus (Günther, 1867)
Strongylura krefftii (Günther, 1866)
Chanos chanos (Forskål, 1775) *tr
Caranx ignobilis (Forskål, 1775) *tr
Lutjanus argentimaculatus (Forskål, 1775) *tr

Non-indigenous species

Astronotus ocellatus (Agassiz, 1831)
Amphilophus citrinellum (Günther, 1864)
Oreochromis mossambicus (Peters, 1852)
Gambusia holbrooki (Girard, 1859)
Poecilia reticulata (Peters, 1859)
Xiphophorus maculatus (Günther, 1866)

Identification references: Munro (1967); Trewavas (1983); Staeck and Linke (1985); Konings (1989); Axelrod (1993); Conkel (1993); Wischnath (1993); Linke and Staeck (1994); Herbert and Peeters (1995); Allen *et al*. (2002)

Table A2 List of aquatic plant species recorded for the Ross River catchment (Webb 1994 and this study)

Submerged and emergent plants

Hydrilla verticillata (L.f.) Royle	
Ceratophyllum demersum L.	
Potamogeton octandrus Hassk.	
Potamogeton tricarinatus Muell. & Benn.	
Potamogeton crispus L.	
Aponogeton queenslandicus van Bruggen	
Najas tenuifolia R.Br.	
Vallisneria sp. (*spiralis*)	
Nymphoides crenata (F. Muell) Kuntze	
Nymphoides indica (L.) Kuntze	
Limnophila indica (L.) Druce	
Nelumbo nucifera Gaertn.	ti
Nymphaea macrosperma Merr. & L.M. Perry	
N. violacea Lehm.	
N. capensis Thunb.	ini
N. mexicana	ini
Utricularia gibba (*exoleta*) (L.)	
Utricularia sp. *(gibba)* (L.)	
Myriophyllum sp.	
Cabomba caroliniana A. Gray	ini
Sagittaria montevidensis Cham. & Schltdl.	ini
Ottelia alismoides (L.) Pers.	

Floating plants

Azolla pinnata R. Br.	
Salvinia molesta D. Mitch.	ini
Eichornia crassipes (C. Mart.) Solms-Laub	ini
Pistia stratiotes L.	ini

ti: translocated indigenous
ini: introduced non-indigenous

Identification references: Aston (1973); Queensland Herbarium (1993); Cowie *et al.* (2000); Sainty and Jacobs (2003)

Table A3 MANOVA summary for comparison of water quality parameters among Ross River weirs and Ross Dam, 1994-1996 (data provided by P. Mockeridge, Citywater Laboratory, Townsville City Council)

MANOVA 1994

	Wilk's Lambda	F	df	p	Eta
	0.042	15.024	14, 54	<0.001	0.796

		p_{crit}: 0.007	Tukey Test (Means comparison)				
	F	p	Ross Dam		Black Weir		Aplin Weir
Conductivity	114.96	<0.001	243.75	≠	332.08	≠	460.33
pH	21.41	<0.001	7.72	=	7.76	≠	8.28
Turbidity	14.69	<0.001	7.54	=	9.62	≠	1.56
Total N	5.47	0.009	138.50	=	90.00	=	35.20
Phosphorus	1.15	0.328	8.33	=	13.33	=	15.00
Calcium	35.41	<0.001	9.62	≠	13.65	≠	17.05
Magnesium	60.941	<0.001	5.38	≠	7.27	≠	11.03

MANOVA 1995

	Wilk's Lambda	F	df	p	Eta
	0.044	14.499	14, 54	<0.001	0.790

		p_{crit}: 0.007	Tukey Test (Means comparison)				
	F	p	Ross Dam		Black Weir		Aplin Weir
Conductivity	86.597	<0.001	227.67	≠	328.58	≠	423.83
pH	6.476	0.004	7.79	=	7.77	≠	8.20
Turbidity	2.975	0.065	7.99	=	12.52	=	4.57
Total N	4.921	0.013	192.75	=	38.75	=	59.42
Phosphorus	5.469	0.009	1.92	=	20.33	=	13.75
Calcium	44.687	<0.001	13.59	≠	5.47	≠	18.77
Magnesium	85.896	<0.001	6.31	≠	7.38	≠	10.75

MANOVA 1996

	Wilk's Lambda	F	df	p	Eta
	0.146	6.242	14, 54	<0.001	0.618

		p_{crit}: 0.007	Tukey Test (Means comparison)				
	F	p	Ross Dam		Black Weir		Aplin Weir
Conductivity	35.000	<0.001	154.08	=	194.67	≠	299.00
pH	7.810	0.002	7.65	=	7.66	≠	8.12
Turbidity	11.049	<0.001	8.21	=	7.82	≠	2.79
Total N	5.998	0.006	195.42	≠	50.42	=	63.75
Phosphorus	2.856	0.072	7.50	=	12.08	=	12.08
Calcium	28.518	<0.001	9.92	=	10.54	≠	15.72
Magnesium	10.798	<0.001	4.34	=	4.68	≠	9.62

Table A4 MANOVA summary for comparison of water quality parameters among Ross River weirs and Ross Dam, 1997-1999 (data provided by P. Mockeridge, Citywater Laboratory, Townsville City Council)

MANOVA 1997

	Wilk's Lambda	F	df	p	Eta
	0.241	3.994	14, 54	<0.001	0.509

		p_{crit}: 0.007	Tukey Test (Means comparison)				
	F	p	Ross Dam		Black Weir		Aplin Weir
Conductivity	5.621	0.008	100.83	=	160.83	=	168.75
pH	3.667	0.036	6.90	=	7.33	=	7.35
Turbidity	1.666	0.204	20.52	=	7.82	=	8.52
Total N	26.041	0.000	172.17	≠	32.67	=	41.74
Phosphorus	0.222	0.802	10.83	=	10.83	=	10.04
Calcium	12.524	<0.001	5.26	≠	9.48	=	10.17
Magnesium	4.697	0.016	2.69	=	4.09	=	4.30

MANOVA 1998

	Wilk's Lambda	F	df	p	Eta
	0.479	1.714	14, 54	0.080	0.308

		p_{crit}: 0.007	Tukey Test (Means comparison)				
	F	p	Ross Dam		Black Weir		Aplin Weir
Conductivity	6.742	0.004	74.9	≠	125.00	=	102.49
pH	5.820	0.007	7.03	≠	7.40	=	7.17
Turbidity	0.047	0.954	21.53	=	17.82	=	17.36
Total N	0.592	0.559	87.08	=	58.42	=	88.67
Phosphorus	0.044	0.957	15.67	=	15.16	=	14.17
Calcium	9.383	0.001	2.60	≠	6.52	=	4.63
Magnesium	5.404	0.009	1.92	≠	3.29	=	2.68

MANOVA 1999

	Wilk's Lambda	F	df	p	Eta
	0.229	4.203	14, 54	<0.001	0.521

		p_{crit}: 0.007	Tukey Test (Means comparison)				
	F	p	Ross Dam		Black Weir		Aplin Weir
Conductivity	12.70	<0.001	82.83	≠	159.66	=	133.51
pH	1.784	0.184	7.35	=	7.53	=	7.38
Turbidity	11.481	0.000*	4.2	≠	25.35	≠	7.62
Total N	3.290	0.050	82.58	=	37.83	=	61.75
Phosphorus	0.336	0.717	11.33	=	13.32	=	11.67
Calcium	13.365	<0.001	2.62	≠	6.91	=	5.31
Magnesium	10.184	<0.001	1.95	≠	3.74	=	3.17

Table A5 MANOVA summary for comparison of water quality parameters among Ross River weirs and Ross Dam, 2000 (data provided by P. Mockeridge, Citywater Laboratory, Townsville City Council)

MANOVA 2000					
	Wilk's Lambda	**F**	**df**	**p**	**Eta**
	0.445	1.927	14, 54	0.044	0.333

		p_{crit}: 0.007	**Tukey Test (Means comparison)**				
	F	**p**	**Ross Dam**		**Black Weir**		**Aplin Weir**
Conductivity	6.133	0.005	91.08	≠	165.42	=	138.92
pH	1.173	0.322	7.39	=	7.55	=	7.42
Turbidity	1.060	0.358	7.20	=	10.44	=	6.83
Total N	0.363	0.698	63.92	=	45.83	=	51.92
Phosphorus	0.216	0.807	11.67	=	11.67	=	10.83
Calcium	10.382	<0.001	3.75	≠	8.65	=	6.51
Magnesium	8.919	0.001	1.97	≠	4.25	=	3.41

Table A6 Water Quality Guidelines for Queensland fresh waters (after Commonwealth Government of Australia (2001))

Parameter	**Method**	**Good**	**Fair**	**Poor**
Conductivity (μS/cm)	Mean	<500	500-1500	>1500
Turbidity (NTU)	Median	<5	5-50	>50
Total N (mg/L)	Median	<0.375	0.375-0.75	>0.75
Total Ph (mg/L)	Median	<0.05	0.05-0.5	>0.5
pH	% months outside range 6.5-9	<10%	10-25%	>25%

Table A7 Water quality assessment for Ross Dam, Aplin and Black weirs, 1994-2000, using guidelines for Queensland fresh waters in Table A6

	Ross Dam		**Black weir**		**Aplin weir**	
Conductivity	139.31	Good	209.42	Good	246.69	Good
Turbidity	5.75	Fair	9.1	Fair	2.7	Good
Total N.	0.98	Poor	0.34	Good	0.40	Fair
Total P.	0.10	Fair	0.10	Fair	0.10	Fair
pH	2.8	Good	0.0	Good	1.4	Good

Table A8 Trophic status criteria and standards for freshwater bodies (after King 1979)

Criteria	**Trophic status**		
	Oligotrophic	**Mesotrophic**	**Eutrophic**
Total Phosphorus (μg/l)	<5	5-30	30-1,000
Total Nitrogen (μg/l)	250	250-1,000	1000-10,000

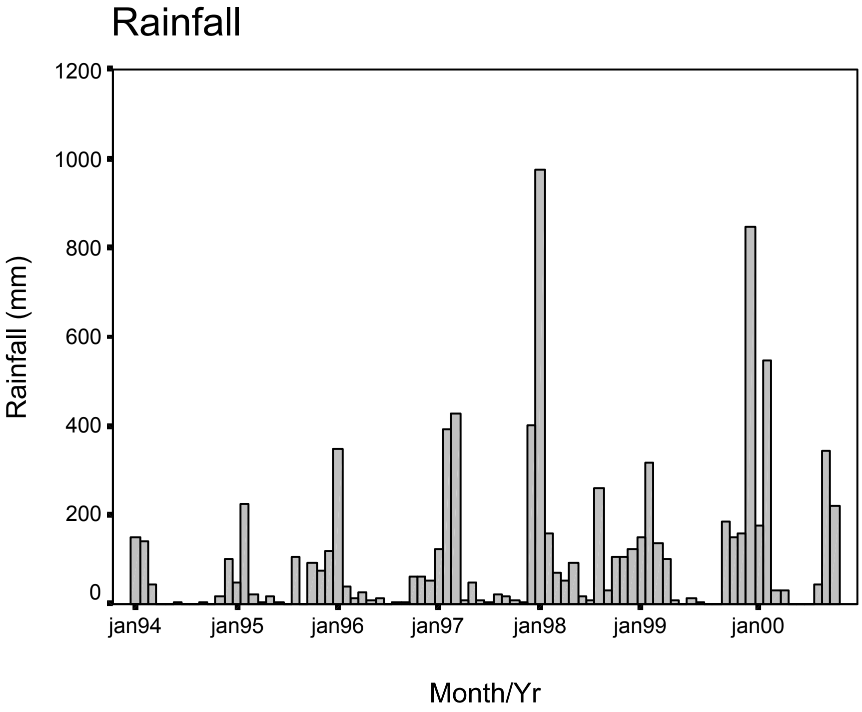


Figure A1 Townsville monthly rainfall data, 1994-2000

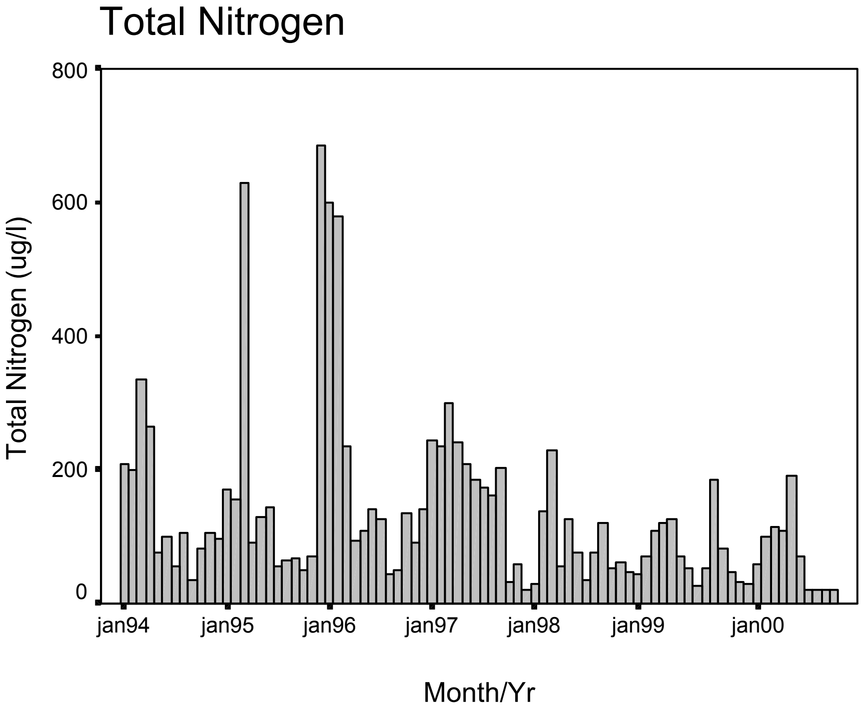


Figure A2 Ross Dam Monthly Total Nitrogen, 1994-2000

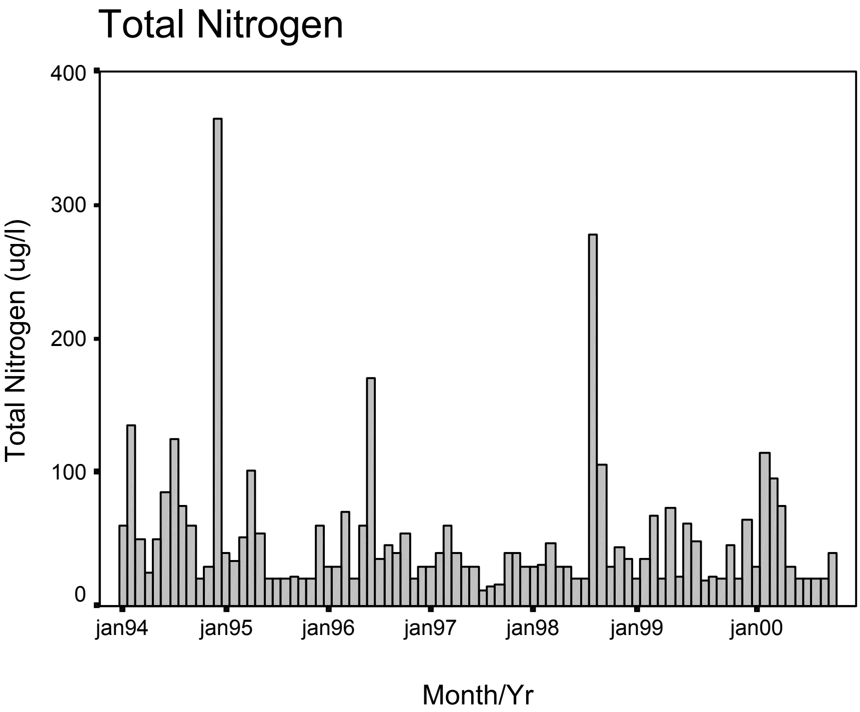


Figure A3 Black Weir, monthly Total Nitrogen, 1994-2000

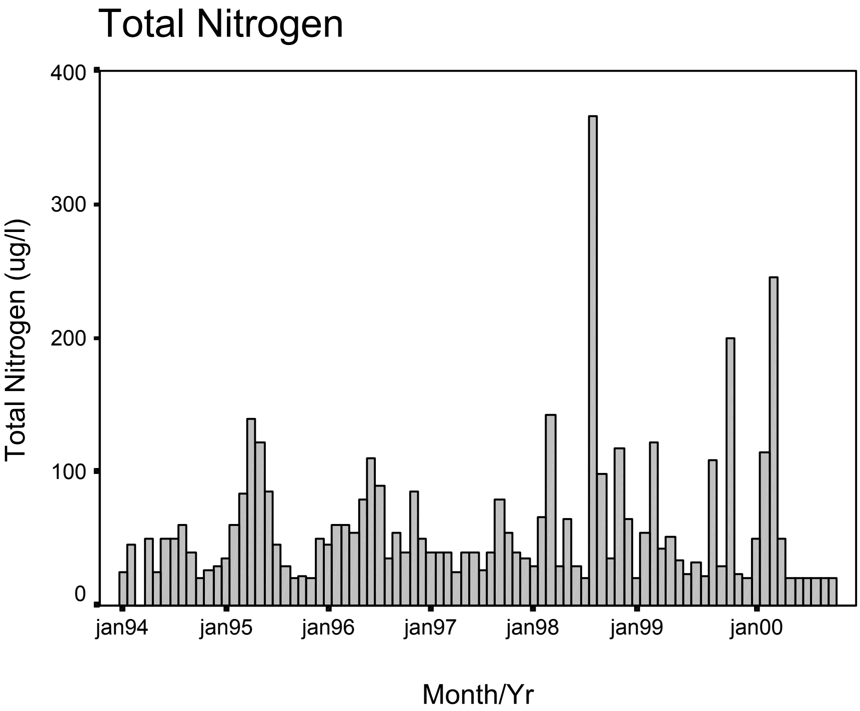


Figure A4 Aplin Weir, monthly Total Nitrogen, 1994-2000

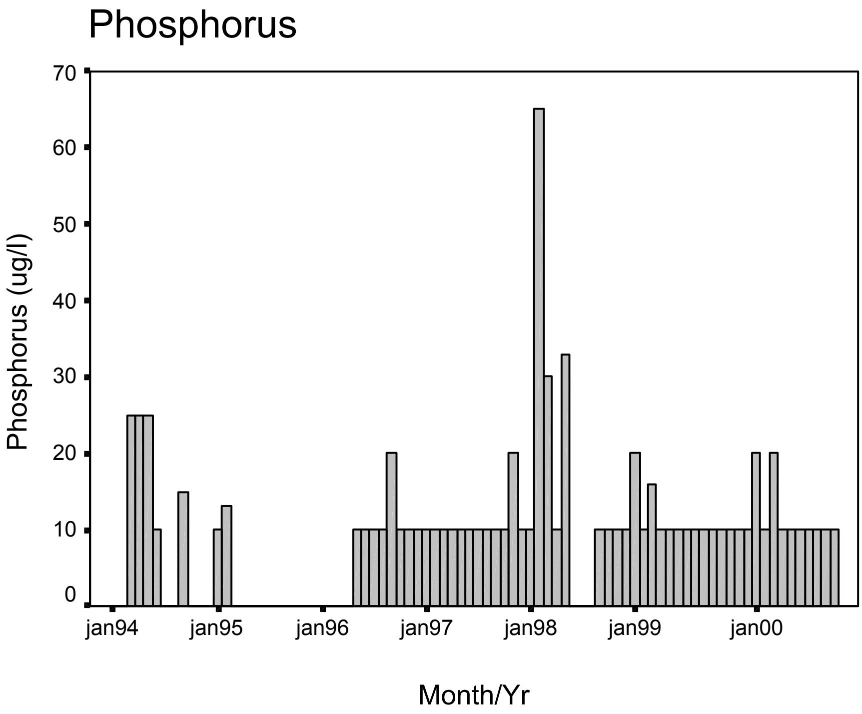


Figure A5 Ross Dam, monthly Phosphorus, 1994-2000

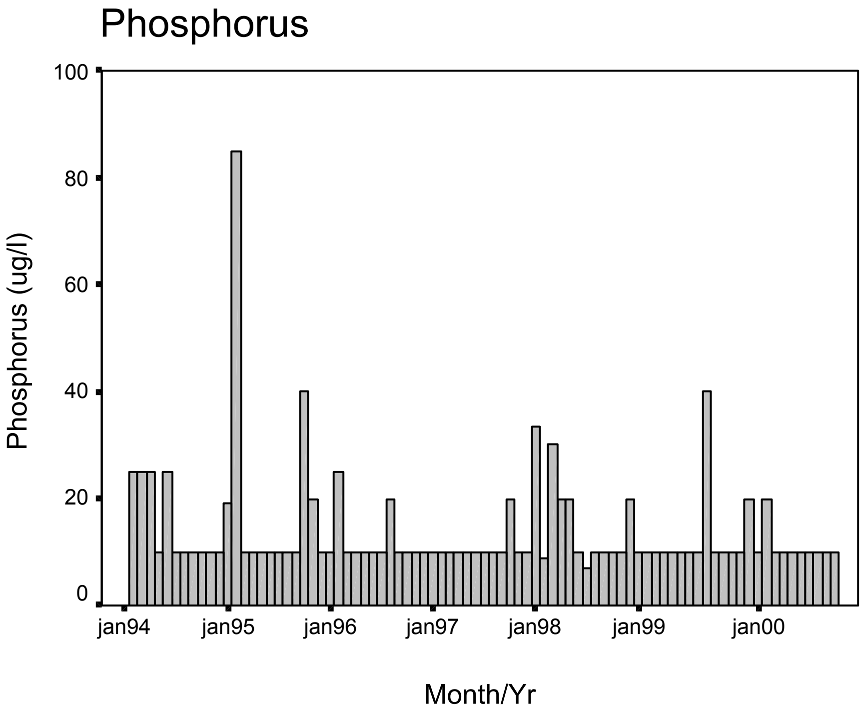


Figure A6 Black Weir, monthly Phosphorus, 1994-2000

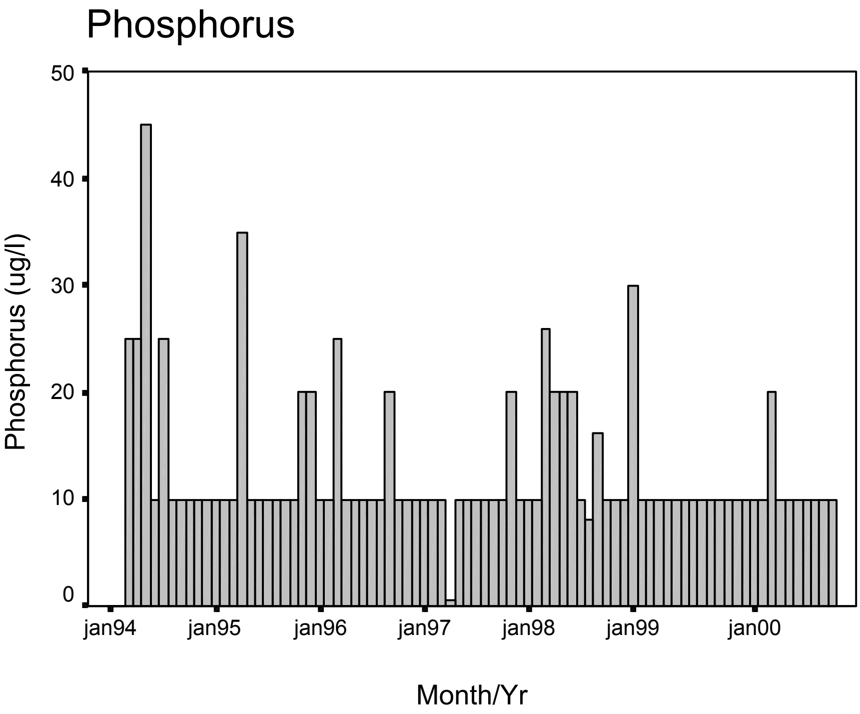


Figure A7 Aplin Weir, monthly Phosphorus, 1994-2000

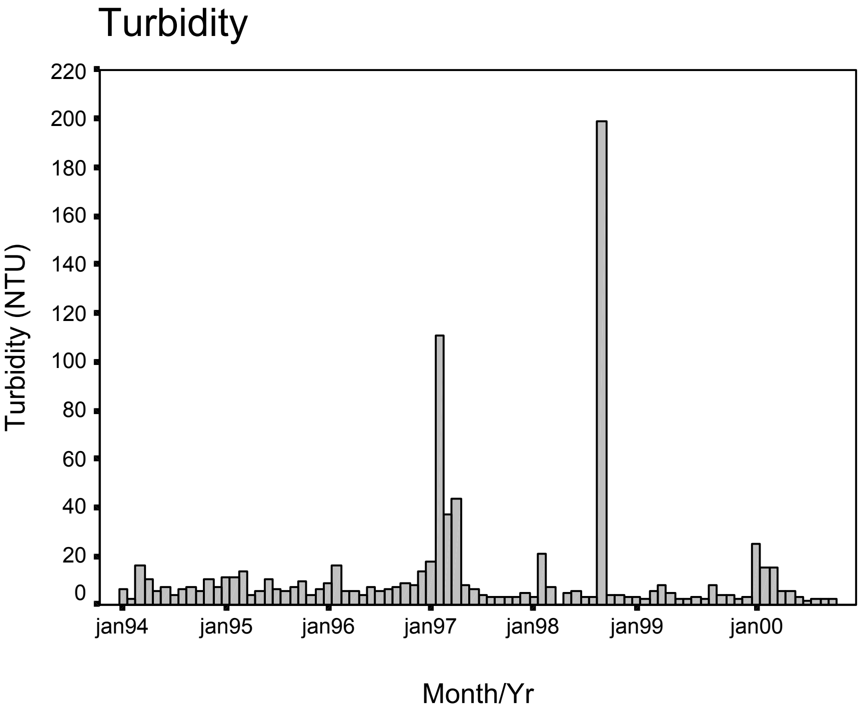


Figure A8 Ross Dam, monthly Turbidity, 1994-2000

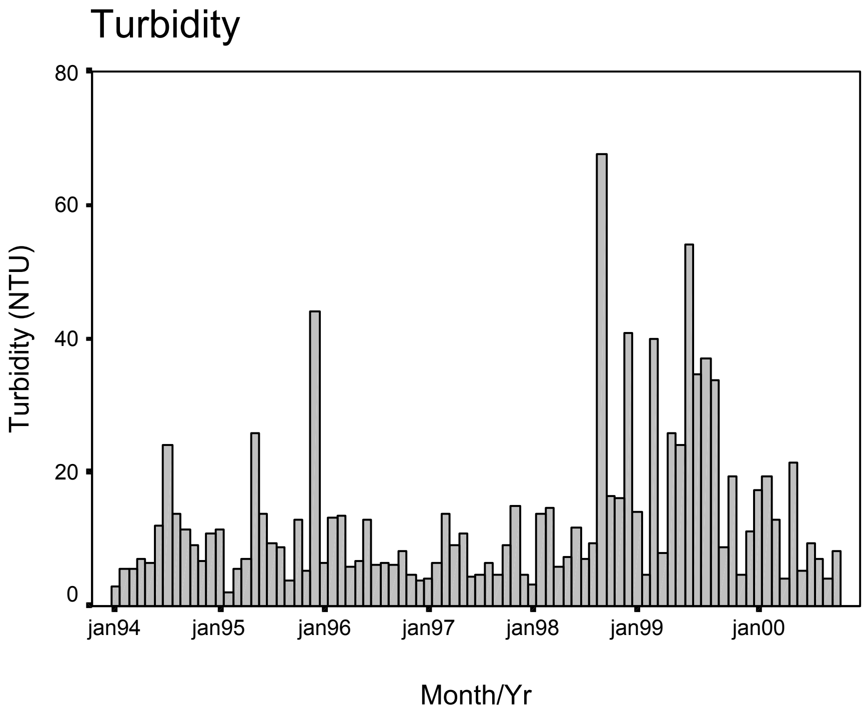


Figure A9 Black Weir, monthly Turbidity, 1994-2000

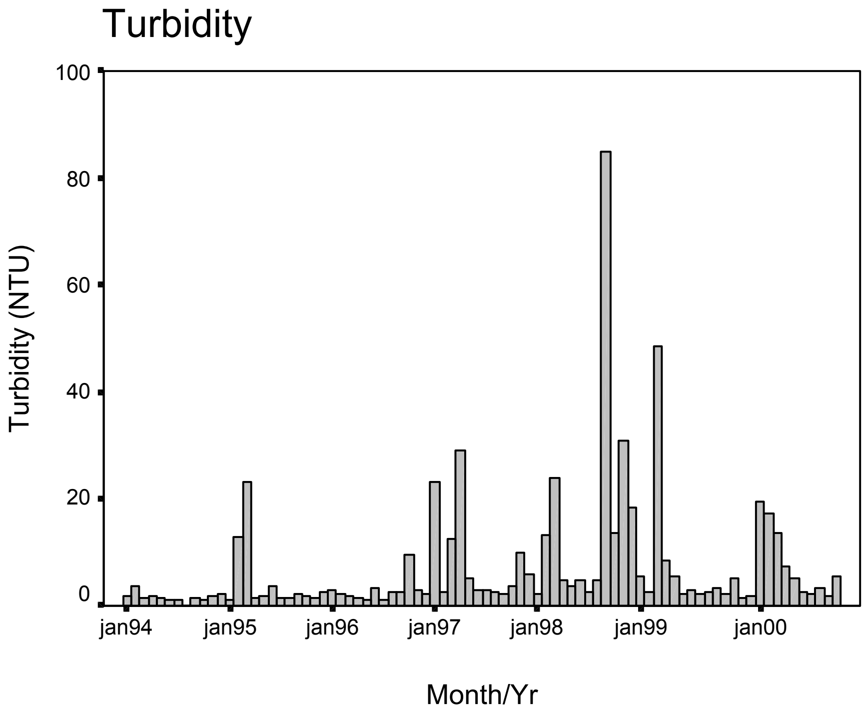


Figure A10 Aplin Weir, monthly Turbidity, 1994-2000

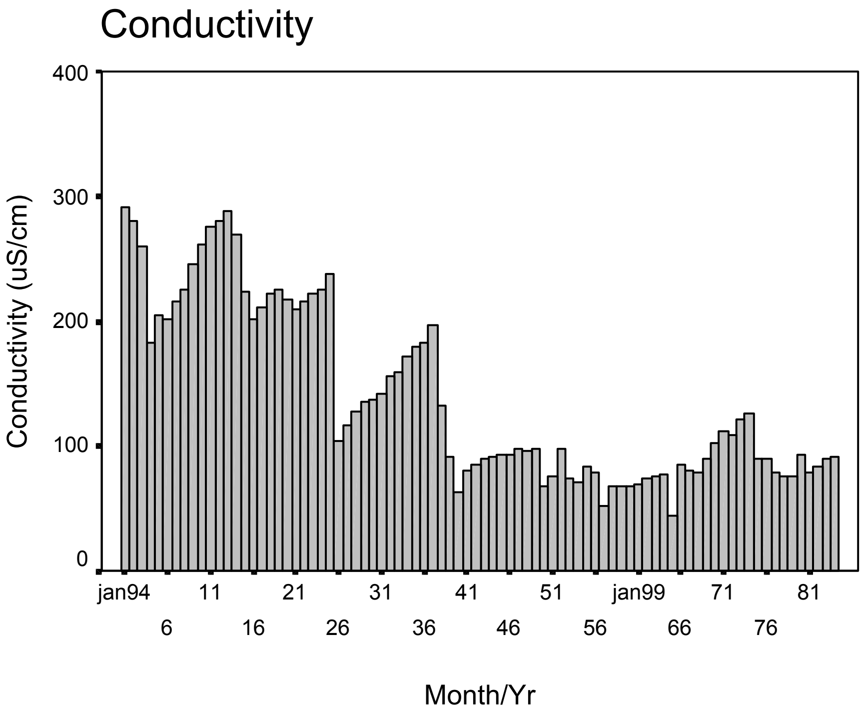


Figure A11 Ross Dam, monthly Conductivity, 1994-2000

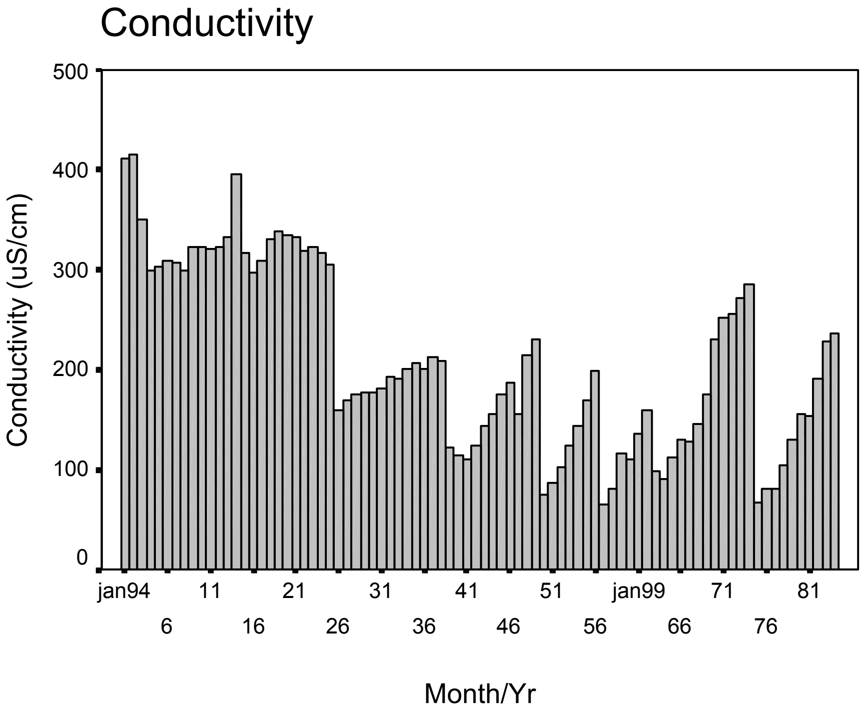


Figure A12 Black Weir, monthly conductivity, 1994-2000

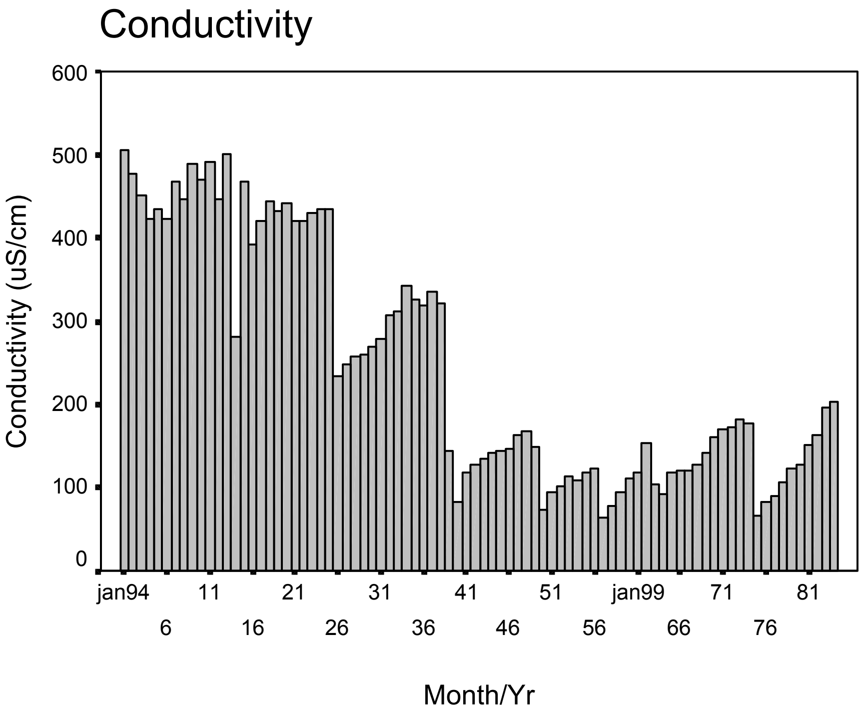


Figure A13 Aplin Weir, monthly Conductivity, 1994-2000

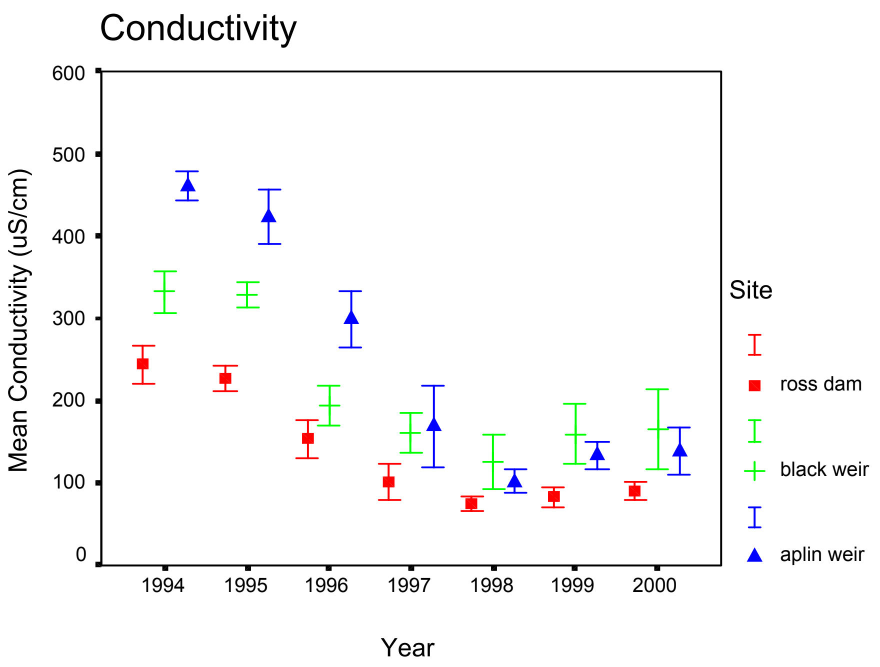


Figure A14 Changes in Conductivity: Ross Dam, Aplin and Black Weir, 1994-2000

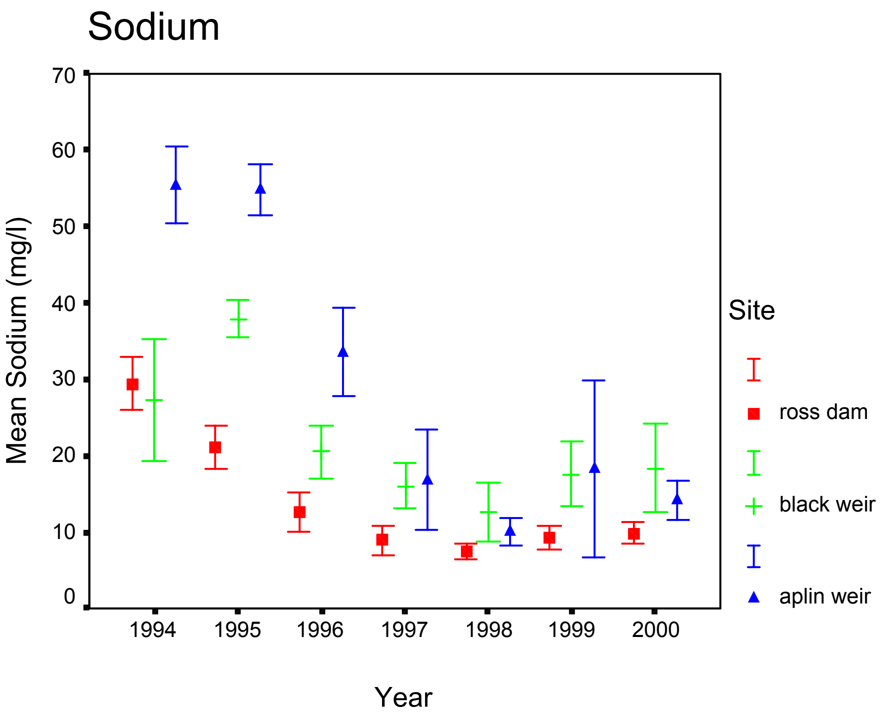


Figure A15 Changes in Sodium: Ross Dam, Aplin and Black Weir, 1994-2000

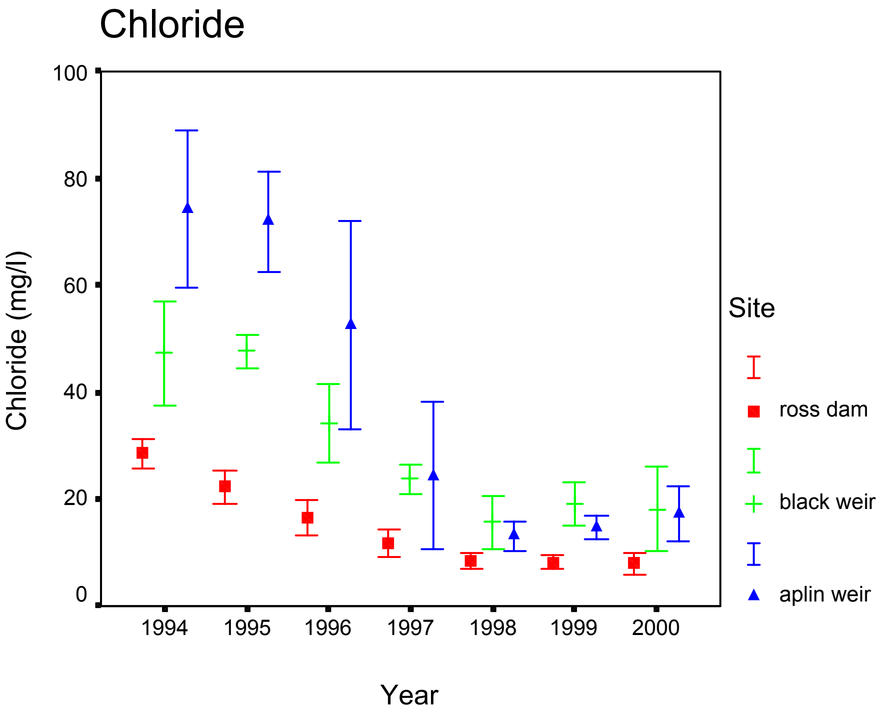


Figure A16 Changes in Chloride: Ross Dam, Aplin and Black Weir, 1994-2000

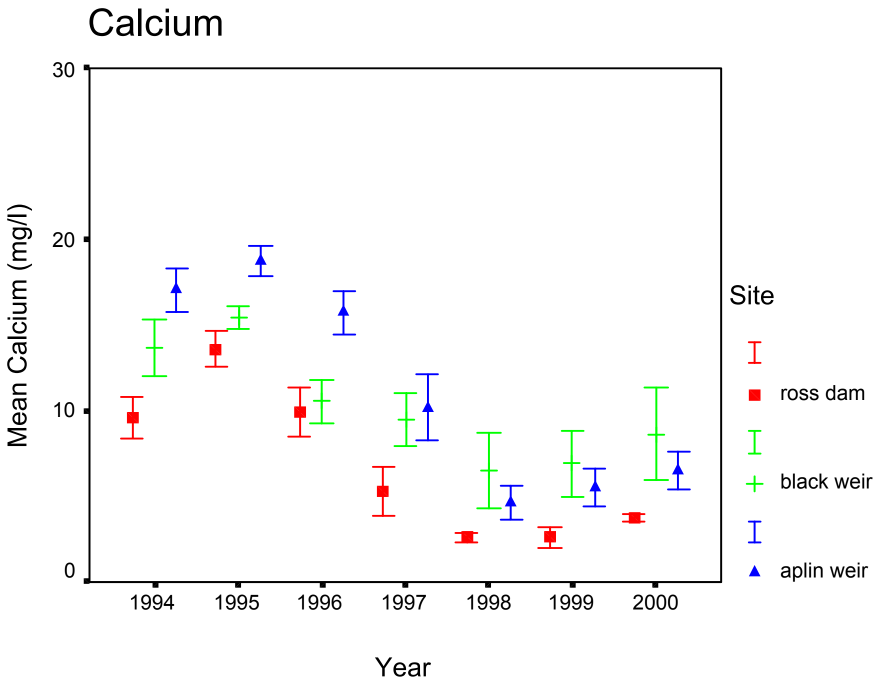


Figure A17 Changes in Calcium: Ross Dam, Aplin and Black Weir, 1994-2000

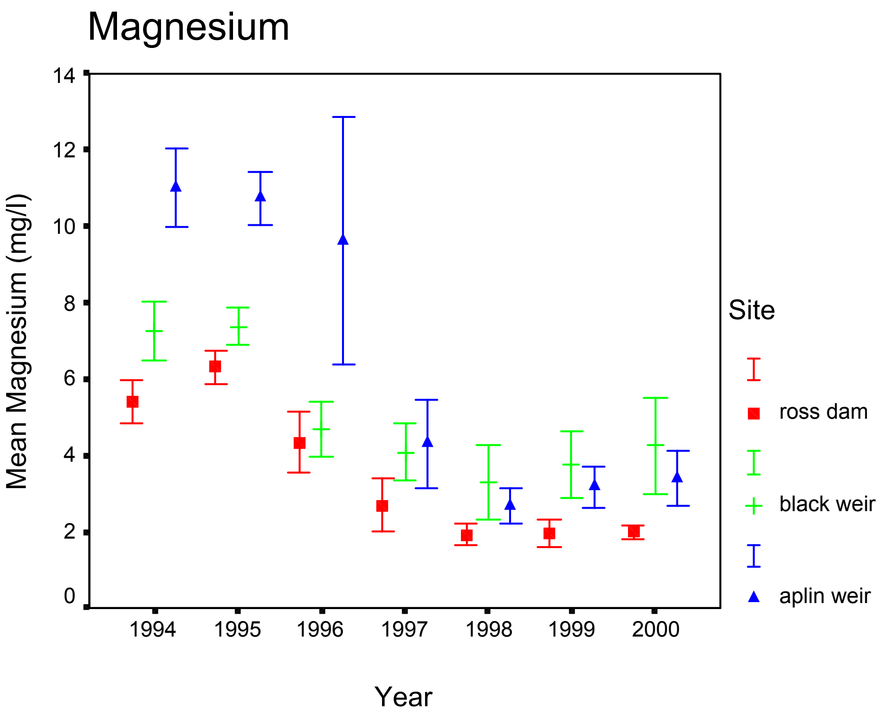


Figure A18 Changes in Magnesium: Ross Dam, Aplin and Black Weir, 1994-2000

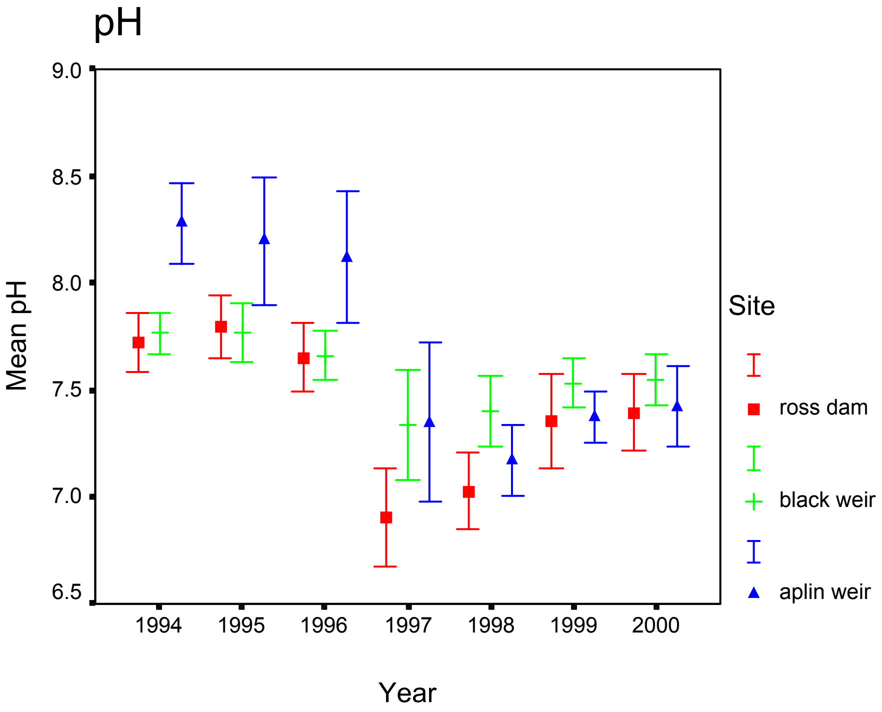


Figure A19 Changes in pH: Ross Dam, Aplin and Black Weir, 1994-2000

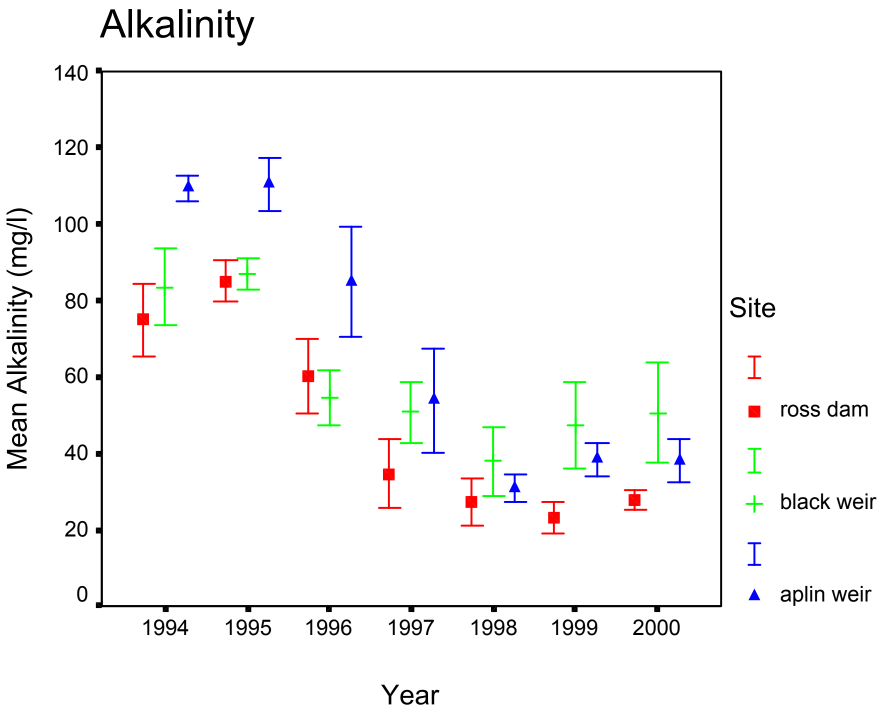


Figure A20 Changes in Alkalinity: Ross Dam, Aplin Weir and Black Weir, 1994-2000

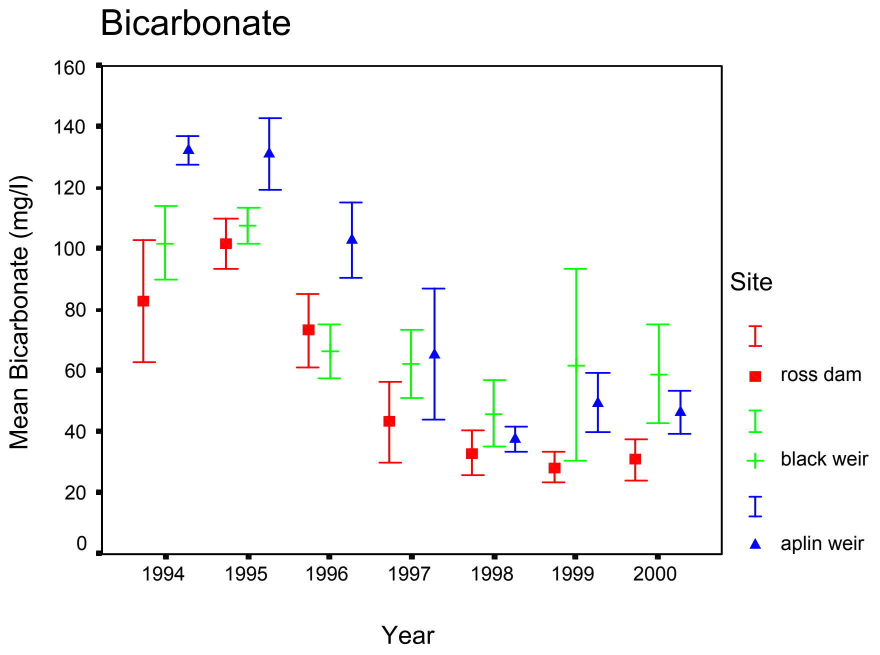


Figure A21 Changes in Bicarbonate: Ross Dam, Aplin and Black Weir, 1994-2000

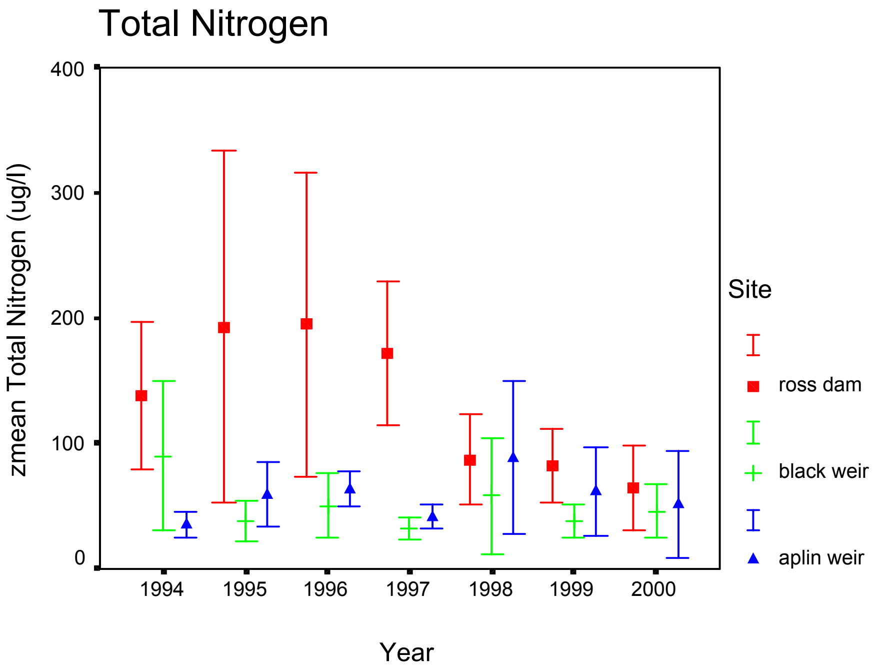


Figure A22 Changes in Total Nitrogen: Ross Dam, Aplin and Black Weir, 1994-2000

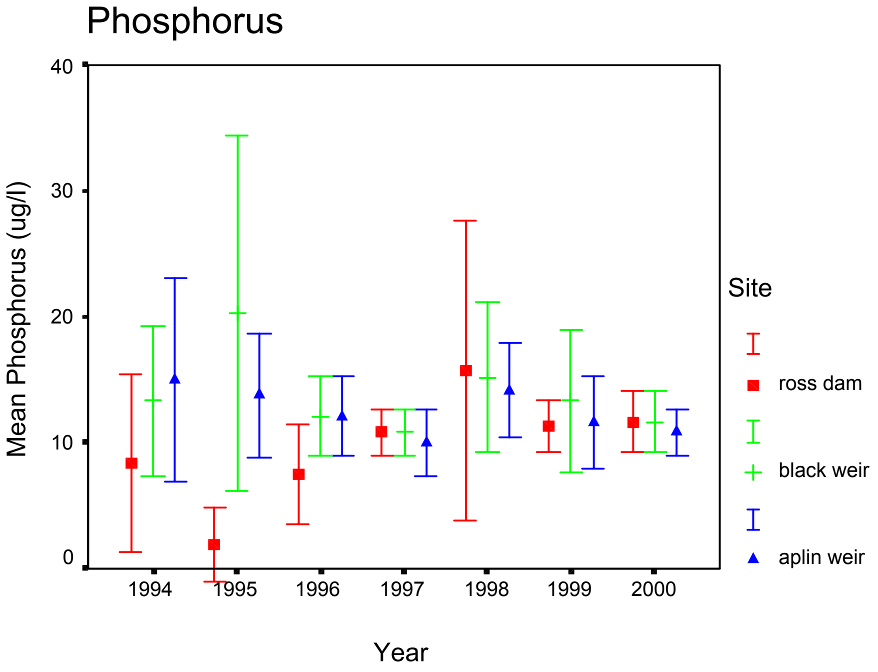


Figure A23 Changes in Phosphorus: Ross Dam, Aplin and Black Weir, 1994-2000

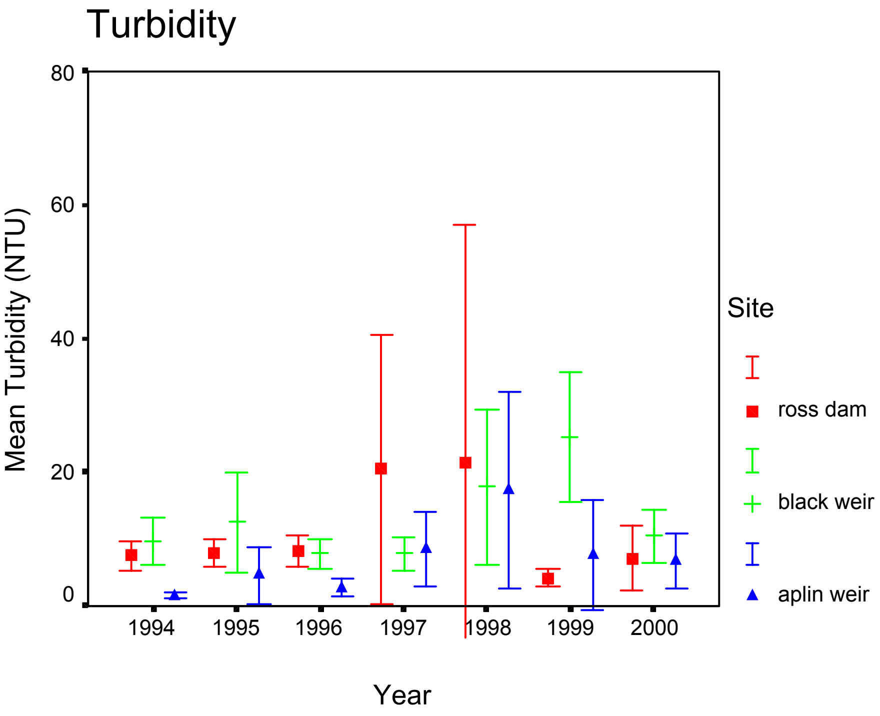


Figure A24 Changes in Turbidity: Ross Dam, Aplin and Black Weir, 1994-2000

APPENDIX B

Table B1 Location and GPS coordinates of freshwater sites sampled for non-indigenous fish species in northern Queensland (p*= sites where fish samples collected for examination of parasite fauna; H=High disturbance habitat; L=Low disturbance habitat – refer to Chapter 6)

	Site	Coordinates	spp	p
1	Daintree River, Daintree	S16.4537º/E145.3148º	Pr	
2	Mossman River, Mossman	S16.4537º/E145.3703º	Pr	
3	Hartleys Creek, Crocodile farm ponds and creek	S16.7417º/E145.5630º	Om	
4	Delaney's Creek, Clifton Beach, north Cairns	S16.7603º/E145.1645º	Xh	* H
5	Deep Creek, north Cairns	S16.7878º/E145.6754º	Pr	* H
6	Half Moon Creek, JCU campus, north Cairns	S16.8173º/E145.6883º	Pr	* H
7	Smithfield Drain, north Cairns	S16.8373º/E145.6962º	Gh	* H
8	Avondale Creek, Smithfield, north Cairns	S16.8458º/E145.6961º	Gh	* H
9	Barron River, Kamerunga, Cairns	S16.8739º/E145.6805º	Tm	*
10	Freshwater Creek, swimming hole	S16.8849º/E145.7035º	Tm	*
11	Brinsmead Drain, Matheson St., Cairns	S16.8965º/E145.7158º	Pr	* H
12	Centenary Lake (fw), Cairns	S16.9034º/E145.7485º	Gh, Xm, Om	*
13	Cairns Civic Theatre, CBD, ornamental pond	S16.9200º/E145.7715º	Xm, Tm, Hg	*
14	Moody's Creek, McGregor St., Cairns	S16.9244º/E145.7374º	Pr	* H
15	Chinaman Creek, Irene St., Cairns	S16.9429º/E145.7287º	Pr	* H
16	Emerald Creek, Atherton Tablelands	S16.9440º/E145.4303º	Pr	* L
17	Barron River, north Mareeba, Atherton Tablelands	S16.9447º/E145.4286º	Pr	* L
18	Gordon Creek, Anderson/Fairview St., Cairns	S16.9665º/E145.7303º	Pr	* H
19	Saw Pit Gully, Forest Gardens, south Cairns	S16.9769º/E145.7350º	Pr, Xh	* H
20	Blackfellow Creek, south Cairns	S17.0136º/E145.7444º	Pr	* L
21	Wright Creek, south Cairns	S17.0401º/E145.7635º	Pr, Om	* L
22	Walkamin Research Station, Atherton Tablelands	S17.1285º/E145.4194º	Pr	* H
23	Mulgrave River, S2, 8km upstream from S1	S17.1466º/E145.8793º	Xm, Tm	*
24	Mulgrave River, S1, Deral, boat ramp, lower reach	S17.2171º/E145.9202º	Tm	*
25	Mazlin Creek, Atherton, Atherton Tablelands	S17.2511º/E145.4683º	Pr	* H
26	Harvey Creek, Bruce Highway	S17.2582º/E145.9234º	Xm	* L
27	Leslie Creek, Peeramon, Atherton Tablelands	S17.3071º/E145.5794º	Om	* H
28	Lower Gwynne Creek, Atherton Tablelands	S17.3119º/E145.0667º	Om	* H
29	"Babinda" Creek 1, Babinda	S17.3400º/E145.9257º	Pr	* H

Pr = Guppy; Om = Mozambique mouthbrooder; Xh = Swordtail; XM = Platy; Hg = Jewel cichlid; Tm = Black-spotted mangrove cihlid; Ao = Oscar; Ac = Midas cichlid; Hs = Green severum; Ar = Green terror; Tt = Three-spotted gourami; Thm = Firemouth cihlid; Hb = Burton's haplochromis; CypA = cyprinid spA

Table B1 contd.

	Site	Coordinates	spp	p
30	"Babinda" Creek 2, Babinda	S17.3483º/E145.9253º	Pr	* H
31	Babinda Creek, Bruce Highway	S17.3604º/E145.9186º	Pr	* H
32	Giddins Creek, Bruce Highway	S17.3620º/E145.9155º	Pr, Xm	* H
33	Upper Gwynne Creek, Atherton Tablelands	S17.3869º/E145.5312º	Pr	* H
34	Pugh Creek, Miriwinni, Bruce Highway	S17.4073º/E145.9073º	Xm	
35	Dirran Creek, Atherton Tablelands	S17.4501º/E145.1097º	Pr	* H
36	Victory Creek, 1km above junction with North Johnstone River	S17.4850º/E146.0002º	Tm	
37	North Johnstone River, near Polly Creek	S17.4940º/E146.0342º	Tm	*
38	Johnstone River, Crocodile Farm ponds, Flying Fish Point	S17.5039º/E146.0451º	Pr, Xh	
39	Small unid. drain next to Johnstone River Crocodile farm	S17.5055º/E146.0445º	Pr	
40	South Johnstone River, lower reach	S17.5477º /E146.0276º	Tm	*
41	Gracy Creek, Wangan	S17.5635º/E146.0261º	Tm	
42	Scheu Creek, Wangan	S17.5811º/E146.0040º	Pr	
43	Moresby Creek, Mourilyan	S17.5832º/E146.0400º	Pr, Xm	* H
44	South Johnstone River, South Johnstone	S17.5993º/E145.9977º	Pr, Tm	
45	South Johnstone River, Sugar Mill	S17.6066º/E145.9895º	Pr, Tm, Xm	
46	Whitfield Creek, Kennedy, Bruce Highway	S18.2030º/E1459532º	Xm	
47	Ripple Creek, north Ingham, Bruce Highway	S18.5836º/E146.2002º	Gh	
48	Drain, Fairford Street, Ingham	S18.6501º/E146.1478º	Gh, Pr	* H
49	Upper Palm Creek, Eleanor Street, Ingham	S18.6513º/E146.1613º	Gh	* H
50	Sandy Waterhole (via Lottery Creek), Ingham	S18.6570º/E146.1573º	Gh	* H
51	Palm Creek Drain, Cooper/Dutton St., Ingham	S18.6573º/E146.1652º	Gh, Pr	* H
52	Drain, Bruce Highway, s. Ingham	S18.8071º/E146.1526º	Gh	* H
53	Little Crystal Creek, floodway	S18.9682º/E146.2902º	Pr	* L
54	Serpentine Lagoon, Woodstock	S19.1009º/E146.8921º	Gh	* H
55	Leichhardt Creek, Bruce Highway	S19.12444/E146.49251	Om	
56	Christmas Creek, Bruce Highway	S19.13618/E146.50510	Om	
57	Sleeper Log Creek, Bruce Highway	S19.14407/E146.51573	Om	
57	Gustav Creek, Magnetic Island	S19.1529º/E146.0462º	Pr	
58	Bluewater Creek, Bruce Highway	S19.1761º/E146.5527º	Om	
59	Healy Creek, Bruce Highway	S19.2091º/E146.0907º	Om	

Pr = Guppy; Om = Mozambique mouthbrooder; Xh = Swordtail; XM = Platy; Hg = Jewel cichlid; Tm = Black-spotted mangrove cihlid; Ao = Oscar; Ac = Midas cichlid; Hs = Green severum; Ar = Green terror; Tt = Three-spotted gourami; Thm = Firemouth cihlid; Hb = Burton's haplochromis; CypA = cyprinid spA

Table B1 contd.

	Site	Coordinates	spp	p
60	Rowes Bay Golf Course, water hazard, Townsville	S19.2254º/E146..7736º	Gh, Om	* H
61	Black River, Bruce Highway, Thuringowa	S19.2276º/E146.6289º	Gh, Pr, Om	* L
62	Blakey's Crossing wetland, Ingham Road, Townsville	S19.2624º/E146.7579º	Gh, Xm, Om	* H
63	Upper Louisa Creek, Mt Louisa, Townsville	S19.2724º/E146.7543º	Gh, Xm, Om	* H
64	Pee Wee Creek, Banfield Dr., Townsville	S19.2839º/E146.7504º	Gh, Xm, Om	
65	Bohle River, Shaw Rd. floodway, Thuringowa.	S19.2902º/E146.7110º	Gh, Om, Ao	* H
66	Anderson Park Lake, Mundingburra, Townsville	S19.2915º/E146.7866º	Gh, Om, An	* H
67	Stormwater drain, below Willow Garden Lake, Thuringowa	S19.3021º/E146.7187º	Gh, Om	* H
68	Ross River, Aplin Weir, above barrage, Townsville	S19.3038º/E146.7802º	Gh, Om, Ao, Ac, Hs, Ar	* L
69	Ross River, Aplin Weir, Bush Garden, Townsville	S19.3047º/E146.7778º	Gh,Xm,Om, Ac	* H
70	Ross River, Aplin Weir, Bush Garden, S2, Townsville	S19.3064º/E146.7778º	Gh, Om	
71	Willow Garden Lake, Hibiscus Village, Thuringowa	S19.3066º/E146.7153º	Gh, Om	* H
72	Ross River, Aplin Weir, Charles Barton Bridge, Townsville	S19.3070º/E146.7612º	Gh, Om, Ao, Ac	* L
73	Ross River, Aplin Weir, Palmetum, Annandale, Townsville	S19.3071º/E146.7653º	Gh, Pr, Om, Ao, Ac, An, Hb	* L
74	Campus Creek, junction with Ross River, Aplin Weir, Townsville	S19.3093º/E146.7657º	Gh, Xm, Om, Ac, Tt	* H
75	"Cranbrook" Creek, Alice St., Cranbrook, Townsville	S19.3106º/E146.7547º	Gh, Pr, Xm, Xh, Om	* H
76	Campus Creek, Palmetum Gardens, Twnsville	S19.3119º/E146.7662º	Gh, Xm, Om Thm An, Hb	* H
77	"Lavarack" Ck. 3, McArthur Dr., Annandale	S19.3126º/E146.7863º	Gh, Xm, Om	* H
78	Palmetum Lake, Douglas, Townsville	S19.3135º/E146.7635º	Gh, Xm, Om, Ac	* H
79	Scrubby Creek, Hervey Range Road, Thuringowa	S19.3154º/E146.0876º	Gh, Pr, Om	* L
80	Alice River, Hervey Range Road, Thuringowa	S19.3157º/E146.0962º	Pr, Gh, Om	* L
81	Ross River, Black Weir, 50m above barrage, Townsville	S19.3161º/E146.7354º	Om	* L

Pr = Guppy; Om = Mozambique mouthbrooder; Xh = Swordtail; XM = Platy; Hg = Jewel cichlid; Tm = Black-spotted mangrove cihlid; Ao = Oscar; Ac = Midas cichlid; Hs = Green severum; Ar = Green terror; Tt = Three-spotted gourami; Thm = Firemouth cihlid; Hb = Burton's haplochromis; CypA = cyprinid spA

Table B1 contd.

	Site	Coordinates	spp	p
82	Canal Creek, Hervey Range Road, Thuringowa	S19.3164º/E146.5761º	Pr, Gh, Om	* L
83	Ross River, Aplin Weir, below Gleeson's barrage, Townsville	S19.3169º/E146.7454º	Gh, Om, Ao, Ac, CypA	* L
84	Regatta Creek, Riverside Gardens, S1, Townsville	S19.3172º/E146.7518º	Gh, Om	* H
85	Regatta Ck., Riverside Boulevard S2, Riverside Gardens, Townsville	S19.3176º/E146.7518º	Gh, Xm, Om	* H
86	"Lavarack" Ck. 4, University Drive, Townsville	S19.3176º/E146.7796º	Gh, Xm, Om	
87	Stormwater Drain, Walkabout, Wulguru, Townsville	S19.3178º/E146.8117º	Gh, Om	
88	Ross River, Gleeson's Weir; immediately above barrage	S19.3181/E146.7452	Cyp., Om	
89	"Lavarack" Ck. 2, University Drive, Townsville	S19.3181º/E146.7994º	Gh, Om	
90	Bohle River, Hervey Range Road, Thuringowa	S19.3183º/E146.7026º	Gh, Om	
91	"Lavarack" Ck. 1, University Drive, Twnsville	S19.3187º /E1468229º	Gh, Om	
92	Log Creek, Hervey Range Road, Thuringowa	S19.3201º/E146.5427º	Gh, Pr, Om	
93	Stuart Creek, Bruce Highway	S19.3221º/E146.8375º	Gh, Om	
94	Campus Creek, JCU Campus, Twnsville.	S19.3450º/E146.7629º	Gh, Xm, Om	
95	Ross River, Black Weir, Loam Island in side channel	S19.3470º/E146.7308º	Gh, Om, Hg,	* H
96	Ross River, Black Weir, Loam Island, main channel	S19.3471º/E146.7308º	Gh, Om, Ao	* L
97	Barrambush Creek, Cungulla, Cape Cleveland	S19.3517º/E147.0359º	Gh, Om	* L
98	Farm Dam (Riley), Riley Road, Cungulla	S19.3529º/E147.0354º	Gh	
99	Stuart Prison Farm dam (Van Tassel Creek), Stuart	S19.3550º/E146.8651º	Gh, Om	
100	Farm Dam (Alvis), Riley Road, Cungulla	S19.3563º/E147.0420º	Gh	
101	Farm Dam (Yallop), Riley Road, Cungulla	S19.3571º/E147.0395º	Gh	
102	Farm Dam (Young), Riley Road, Cungulla	S19.3580º/E147.0404º	Gh, Om	*
103	Farm Dam (Noble), Riley Road, Cungulla	S19.3604º/E147.0387º	Gh, Om	*
104	Farm Dam (Lyons), Riley Road, Cungulla	S19.3620º/E147.0373º	Gh, Om	
105	Ross River, Black Weir, Apex Park, Townsville	S19.3624º/E146.7329º	Gh, Pr, Om	

Pr = Guppy; Om = Mozambique mouthbrooder; Xh = Swordtail; XM = Platy; Hg = Jewel cichlid; Tm = Black-spotted mangrove cihlid; Ao = Oscar; Ac = Midas cichlid; Hs = Green severum; Ar = Green terror; Tt = Three-spotted gourami; Thm = Firemouth cihlid; Hb = Burton's haplochromis; CypA = cyprinid spA

Table B1 contd.

	Site	Coordinates	spp	p
106	Drainage line, Carty Road, Cungulla	S19.3627º/E147.0354º	Gh, Om	
107	Drainage line, Meehan Rd, Cungulla	S19.3665º/E147.0481º	Gh	
108	Drainage line, Meehan Rd, Cungulla	S19.3685º/E145.0481º	Gh	
109	Stoney Creek, Mt Stuart, Townsville	S19.3733º/E146.8360º	Gh, Om	
110	Ross River, Black Weir, 200m below Ross Dam	S19.4090º/E146.7356º	Gh, Om	* L
111	Ross Dam (Stuart), S1, near dam spillway	S19.4124º/E146.7431º	Gh	
112	Ross Dam (Stuart), S2	S19.4169º/E146.7499º	Gh	
113	Alligator Creek, Mt Stuart, below National Park	S19.4259º/E146.9468º	Pr, Om	* L
114	Farm Dam, Oak Valley (Thomas), Braby Rd.	S19.4265º/E146.8127º	Gh, Om	
115	Farm Dam, Oak Valley (Thomas), Braby Rd.	S19.4265º/E146.8120º	Gh, Om	
116	Farm Dam. Oak Valley (Thomas), Braby Rd.	S19.4271º/E146.8133º	Gh, Om	
117	Alligator Creek, Mt Stuart, Nat. Park	S19.4365º/E146.9493º	Pr, Om	
118	Upper Ross River Wetland, Thuringowa.	S19.4572º/E146.7254º	Gh	* H
119	Ross River Dam (Kelso) S1	S19.4584º/E146.7278º	Gh	
120	Cromarty Wetland, southern end	S19.4628º/E147.0498º	Gh	* L
121	St Margaret's Creek, lower reach	S19.4734ºE147.0605º	Gh	* L
122	Ross Dam (Kelso), S2	S19.4746º/E146.7204º	Gh	
123	Sandy Creek	S19.48576º/E146.7113º	Gh	
124	Central Creek	S19.49181º/E146.6928º	Gh	
125	Mountain View Lake, Palm Creek, Bruce Highway.	S19.5070º/E147.0326º	Gh	*
126	Healey's Lagoon, canefield drain, Bruce Hway.	S19.5169º/E147.0459º	Gh	* H
127	Crooked Waterhole, Giru	S19.5178º/E147.0940º	Gh	
128	Kalamia Sugar Mill, Lagoon, Ayr	S19.5183º/E147.4205º	Gh	
129	Gorizia's Lagoon, Sheep Station Creek, Brandon	S19.5247º/E147.3342º	Gh, Tt	* H
130	Gravel Pit, Beach Road, Ayr	S19.5490º/E147.4185º	Gh	
131	Pink Lilly Lagoon, Bruce Highway	S19.5507º/E147.0992º	Gh	
132	Cane Drain, S. Brandon, Bruce Highway	S19.5557º/E147.3684º	Gh	
133	Collinsons Lagoon, Bruce Highway	S19.5585º/E147.2920º	Gh	

Pr = Guppy; Om = Mozambique mouthbrooder; Xh = Swordtail; XM = Platy; Hg = Jewel cichlid; Tm = Black-spotted mangrove cihlid; Ao = Oscar; Ac = Midas cichlid; Hs = Green severum; Ar = Green terror; Tt = Three-spotted gourami; Thm = Firemouth cihlid; Hb = Burton's haplochromis; CypA = cyprinid spA

Table B1 contd.

	Site	Coordinates	Species	p
134	Banister Bog, Bruce Highway	S19.5589º/E147.2628º	Gh	
135	Horseshoe Lagoon, Bruce Highway	S19.5604º/E147.1275º	Gh	
136	Didgeridoo Lagoon, Bruce Highway	S19.5604º/E147.2480º	Gh	
137	Sheep Station Creek 1, Bruce Highway	S19.5607ºE147.0445º	Gh, Tt	
138	Irrigation channel, n. Barrattas, Bruce Highway	S19.5653º/E147.1698º	Gh	
139	West Barratta Creek, Bruce Highway	S19.5677º/E147.2006º	Gh	
140	East Barratta Creek, Bruce Highway	S19.5696º/E147.2219º	Gh	
141	Plantation Creek, Bruce Highway, S. Ayr	S19.5878º/E147.3982º	Gh	
142	Sheep Station Creek, Pearce Bridge, upstream Dick's Bank Lagoon	S19.5974º/E147.3400º	Gh, Tt	* H
143	Plantation Creek S3, S. Ayr	S19.6021º/E147.3632º	Gh	
144	Majors Creek, Woodstock Road	S19.6036º/E146.9095º	Gh, Pl	* L
145	Plantation Creek S2, S. Ayr	S19.6243º/E147.0713º	Gh	
146	Plantation Creek, S4, Dalbeg Road	S19.6380º/E147.3811º	Gh	
147	Artificial Lagoon, Mt Kelly Drive	S19.6534º/E147.0311º	Gh, Tt	
148	Sheep Station Creek, J. Aherne Bridge, Airville	S19.6543º/E147.3416º	Gh, Tt	* H
149	Cane Drain, Mt Kelly Drive	S19.6570º/E147.3306º	Gh, Tt	* H
150	Irrigation Channel, Lena Road	S19.6807º/E147.3141º	Gh, Tt	* H
151	Gum Holes floodway (Barratta Creek)	S19.6812º/E147.1226º	Gh	
152	Cane Drain, (Dowson) Dalbeg Road	S19.6971º/E147.0080º	Gh, Tt	* H
153	Irrigation Channel Dr RB3, Brown Road (below Kratzman)	S19.6987º/E147.2176º	Gh, Tt	
154	Irrigation Channel (Cox), cnr Brown/Pelican Road	S19.6995º/E147.2253º	Gh	
155	The Rocks, Pumping Station, Dalbeg Road	S19.7024º/E147.2927º	Gh	
156	Cane Drain, cnr Shepherd/Dalbeg Road	S19.7284º/E147.2592º	Gh	
157	Irrigation Channel Ba5 from Haughton Pump Station, (Kratzman) Northcote	S19.73112º/E147.2114º	Gh, Tt	
158	Cane Drain, Old Clare Road, Clare	S19.7642º/E147.2294º	Gh	
S1	Lake Tinaroo (Barron River Arm), Atherton Tablelands (AT)	S17.2542º/E145.5468º	Om	

Pr=Guppy; Om = Mozambique mouthbrooder; Xh = Swordtail; XM = Platy; Hg = Jewel cichlid; Tm = Black-spotted mangrove cihlid; Ao = Oscar; Ac = Midas cichlid; Hs = Green severum; Ar = Green terror; Tt = Three-spotted gourami; Thm = Firemouth cihlid; Hb = Burton's haplochromis; CypA = cyprinid spA

Table B1 contd.

	Site	Coordinates	Species	p
S2	Leslie Creek (0.5km above S5), Peeramon, AT	S17.2973º/E145.5420º	Om	
S3	Leslie Creek (50m above Barron River junction) AT	S17.2925ºº/E145.5420º	Om	
S4	Barron River (5km below Leslie Creek, AT	S17.2657º/E145.5422º	Om	
S5	Barron River (between Pink's Bridge and Leslie Creek) AT	S17.2750º/E145.5378º	Om	
S6	Barron River (100m above Picnic Crossing) AT	S17.3055º/E145.5678º	Om	
S7	Gwynne Creek, Peeramon, AT	S17.3055º/E145.5678º	Om	
S8	Gwynne Creek (50m upstream from S19) AT	S17.3098º/E145.5678º	Om	
S9	Gwynne Creek (below waterfall) AT	S17.3107º/E145.5675º	Om	
S10	Landcare Channel (flows into Leslie Creek, Peeramon) AT	S17.3105º/E145.5840º	Om	
S11	Ornamental Pond (ex Clough Creek, Matthews) Peeramon, AT*	S17.3103º/E145.5818º	Om	
S12	Leslie Creek (100m upstream Landcare Channel) AT	S17.3073º/E145.5835º	Om	
S13	Kenney Creek (100m below Foxwell culvert) AT	S17.3195º/E145.5810º	Om	
S14	Collins Dam No.2, Kenney Creek, AT	S17.3200º/E145.5765º	Pr, Om, Tm	
S15	Kenney Creek (200m upstream S32) AT	S17.3235º/E145.5812º	Om	
S16	Drainage Channel (Collins Dam No.1) Kenney Creek, AT	S17.3247º/E145.5828º	Om	
S17	Drainage Channel (200m upstream S16)	S17.3290º/E145.5803º	Om	
S18	McKeague's Dam (tributary of Kenney Creek) AT	S17.3315º/E145.5862º	Om	
S19	Peeters Dam (Kenney Creek) AT	S17.3358º/E145.5752º	Om	
S20	Collins Dam No.2, location 2	S17.3150º/E145.5752º	Pr, Om, Tm	
S21	Collins Dam No. 1 (Kenney Creek) AT	S17.3150º/E145.5813º	Pr, Om, Tm	
T1	Wadda Creek, Johnstone River	S17.5976º/E145.8408º	Pr	
T2	Bora Creek, Johnstone River	S17.5529º/E145.8411º	Pr	
T3	Berner Creek, Johnstone River	S17.5617º/E145.8905º	Pr	
T4	Meunbah Creek, Johnstone River	S17.6783º/E145.9723º	Pr	
T5	Mena Creek, Johnstone River	S17.6762º/E145.9571º	Pr	
T6	North Johnstone River	S17.5462º/E145.8592º	Pr	
T7	Nind's Creek, Johnstone River	S17.5691º/E146.0869º	Xm	
T8	Cleminson Creek, AT	S17.3474º/E145.5765º	Pr	

Pr = Guppy; Om = Mozambique mouthbrooder; Xh = Swordtail; XM = Platy; Hg = Jewel cichlid; Tm = Black-spotted mangrove cihlid; Ao = Oscar; Ac = Midas cichlid; Hs = Green severum; Ar = Green terror; Tt = Three-spotted gourami; Thm = Firemouth cihlid; Hb = Burton's haplochromis; CypA = cyprinid spA

Table B1 contd.

	Site	Co-ordinates	Species	p
U1	Jack’s Lagoon, Sheep Station Creek, Lower Burdekin	S19.5009º/E147.3222º	Gh, Tt	
U2	Dick’s Bank Lagoon, Sheep Station Creek	S19.5798º/E147.3454º	Gh, Tt	
U3	Kelly’s Lagoon, Sheep Station Creek	S19.6046º/E147.3273º	Gh, Tt	
U4	Payard’s Lagoon, Sheep Station Creek	S19.6152º/E147.3150º	Gh, Tt	
X1	Daintree River	S16.1927º/E145.2928º	Pr	
X2	Johnstone River	S17.3627º/E145.9276º	Pr	
X3	Johnstone River	S17.5317º/E145.9801º	Tm	
X4	Johnstone River	S17.5066º/E145.9925º	Tm	
X5	Johnstone River	S17.5073º/E146.0020º	Tm	
X6	Johnstone River	S17.5031º/E146.0030º	Tm	
X7	Johnstone River	S17.4928º/E146.0070º	Tm	
X8	Johnstone River	S17.4901º/E146.0040º	Tm	
X9	Johnstone River	S17.4930º/E146.0010º	Tm	
X10	Johnstone River	S17.5858º/E146.0180º	Tm	
X11	Johnstone River	S17.5831º/E146.0217º	Tm	

Pr = Guppy; Om = Mozambique mouthbrooder; Xh = Swordtail; XM = Platy; Hg = Jewel cichlid; Tm = Black-spotted mangrove cihlid; Ao = Oscar; Ac = Midas cichlid; Hs = Green severum; Ar = Green terror; Tt = Three-spotted gourami; Thm = Firemouth cihlid; Hb = Burton’s haplochromis; CypA = cyprinid spA

Reference: SITES S: Webb *et al*. 1996; **SITES T**: D.J. Russell, Northern Fisheries Centre, Cairns, *pers. comm*.; **SITES U**: C. Perna, ACTFR, Townsville, *pers. comm*.; **SITES X**: QFS (2000) Long Term Monitoring Program

Table B2 Non-indigenous fishes recorded (R) or established (E) in Australian fresh waters (*including data from this study)

FAMILY	SPECIES	COMMON NAME	*STATUS*
Belontiidae	*Trichogaster trichopterus*	Three-spot gourami	E
Cichlidae	*Aequidens pulchrus*	Blue acara	R
Cichlidae	*Aequidens rivulatus*	Green terror	R*
Cichlidae	*Amphilophus citrinellum*	Midas cichlid	E
Cichlidae	*Archocentrus nigrofasciatum*	Convict cichlid	R*
Cichlidae	*Archocentrus octofasciatum*	Jack Dempsey	R
Cichlidae	*Astronotus ocellatus*	Oscar	E
Cichlidae	*Geophagus brasiliensis*	Pearl cichlid	R
Cichlidae	*Haplochromis burtoni*	Burton's haplochromis	E*
Cichlidae	*Hemichromis guttatus*	Jewel cichlid	E
Cichlidae	*Nandopsis trimaculatum*	Three spot cichlid	E
Cichlidae	*Oreochromis mossambicus*	Mozambique mouthbrooder	E
Cichlidae	*Thorichthys meeki*	Firemouth cichlid	R*
Cichlidae	*Tilapia mariae*	Black mangrove cichlid	E
Cichlidae	*Tilapia zillii*	Redbelly tilapia	E
Cobitidae	*Misgurnus anguillicaudatus*	Oriental weather loach	E
Cyprinidae	*Carassius auratus*	Goldfish	E
Cyprinidae	*Cyprinus carpio*	European carp	E
Cyprinidae	*Unid. cyprinid sp. A*	"Carp"	R*
Cyprinidae	*Puntius conchonius*	Rosy barb	R
Cyprinidae	*Puntius tetrazona*	Sumatra barb	R
Cyprinidae	*Rutilus rutilus*	Roach	E
Cyprinidae	*Tanichthys albonubes*	White cloud mountain minnow	E
Cyprinidae	*Tinca tinca*	Tench	E
Percidae	*Perca fluviatilis*	Redfin perch	E
Poeciliidae	*Gambusia holbrooki*	Mosquitofish	E
Poeciliidae	*Phalloceros caudimaculatus*	One-spot livebearer	R
Poeciliidae	*Poecilia latipinna*	Sailfin molly	E
Poeciliidae	*Poecilia reticulata*	Guppy	E
Poeciliidae	*Xiphophorus helleri*	Swordtail	E
Poeciliidae	*Xiphophorus maculatus*	Platy	E
Salmonidae	*Onchorhynchus mykiss*	Rainbow trout	E
Salmonidae	*Onchorhynchus tshawytscha*	Chinook salmon	R
Salmonidae	*Salmo salar*	Atlantic salmon	R
Salmonidae	*Salmo trutta*	Brown trout	E
Salmonidae	*Salvelinus fontinalis*	Brook trout	R

Table B3 Numbers of introduced freshwater fish species from selected countries in comparison with Australia, Queensland and northern Queensland (number of established species in parentheses)

	Σ intro.. spp	Characidae (Tetras)	Characidae (others)	Catfish several families	Cyprinidae (Barbs)	Cichlidae	Poeciliidae
Taiwan	38						
Thailand	92						
India	300						
China	109						
Japan	120						
Philippines	38						
Malaysia	54						
Indonesia	20						
Singapore	39						
Sri Lanka	24						
Spain	25						
Portugal	11						
Netherlands	20						
Greece	21						
Italy	49						
France	36						
UK	22						
Netherlands	27						
Germany	18						
South Africa	20						
Turkey	23						
Argentina	11						
Brazil	36						
Colombia	37						
Chile	21						
Peru	17						
Bolivia	11						
Dominican Rep.	16						
Costa Rica	13						
Mexico	55						
USA	194	6 (0)	10 (0)	18 (4)	4 (0)	35 (20)	13 (10)
Florida	103	4 (0)	5 (0)	13 (5)	4 (0)	31 (19)	13 (9)
Hawaii	75	0 (0)	2 (0)	4 (3)	2 (2)	18 (16)	10 (9)
Australia	38	0 (0)	0 (0)	0 (0)	2 (0)	17 (7)	6 (5)
Qld	27	0 (0)	0 (0)	0 (0)	2 (0)	14 (6)	5 (5)
NQld	17	0 (0)	0 (0)	0 (0)	0 (0)	10 (6)	5 (5)

Data from: Contreras and Escalente 1984; Courtenay *et al*. 1984; De Groot 1985; Radtke 1995; De Silva (ed.) 1989; Almaca 1995; Farr-Cox *et al.* 1996; Yuma *et al.* 1998; Kailola *et al*. 1999; Economidis et al. 2000; Elvira and Almodovar 2001; de Magalhaes *et al*. 2002; USGS Nonindigenous Aquatic Species Database 2003; Fishbase 2003; Webb (this study); T. Rayner, JCUNQ, School of Tropical Biology, *pers.comm*.

Figure B1 Convict cichlid, *Archocentrus nigrofasciatum* (adult TL: 10cm)

Figure B2 Firemouth cichlid, *Thorichthys meeki*, (adult, TL: 6cm)

Figure B3 Green severum, *Heros severus* (Juvenile, TL: 5cm)

Figure B4 Green terror, *Aequidens rivulatus* (Juvenile, TL: 9cm)

Figure B5 Black-spotted mangrove cichlid, *Tilapia mariae* (Juvenile, TL: 8cm)

Figure B6 Mozambique mouthbrooder, *Oreochromis mossambicus* (Juvenile, TL: 5cm)

Figure B7 Jewel cichlid, *Hemichromis guttatus* (Adult, TL: 6cm)

Figure B8 Burton's haplochromis, *Haplochromis burtoni* (Adult, TL: 6cm)

Figure B9 Oscar, *Astronotus ocellatus* (Juvenile, TL: 12cm)

Figure B10 Midas cichlid, *Amphilophus citrinellum* (Red morph, Adult, TL: 23cm)

Figure B11 Mosquitofish, *Gambusia holbrooki* (Adult female, TL: 4cm)

Figure B12 Platy, *Xiphophorus maculatus* (Adult female, TL: 3cm)

Figure B13 Swordtail, *Xiphophorus helleri* (Adult male, TL: 6cm)

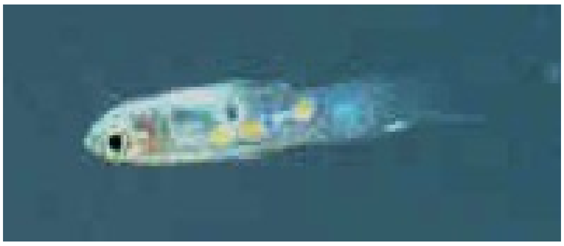


Figure B14 Guppy, *Poeclia reticulata* (Adult male, TL: 1.2cm)

Figure B15 Guppy, *Poecilia reticulata* (Adult female, TL: 1.8cm)

Figure B16 Sailfin molly, *Poecilia latipinna* (Adult, TL: 5cm)

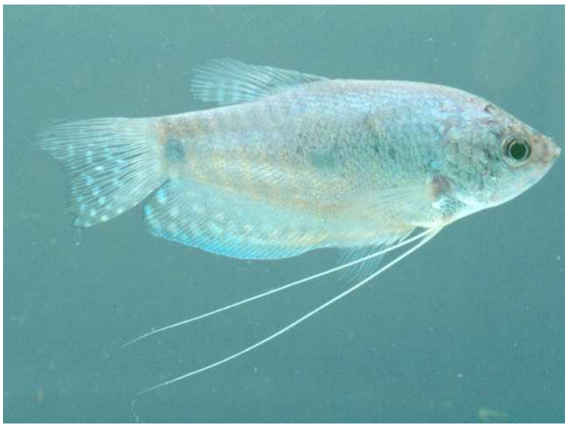


Figure B17 Three-spot gourami, *Trichogaster trichopterus* (Adult, TL: 7cm)

Figure B18 Unidentified cyprinid sp. A (Adult, TL: approx. 45cm)

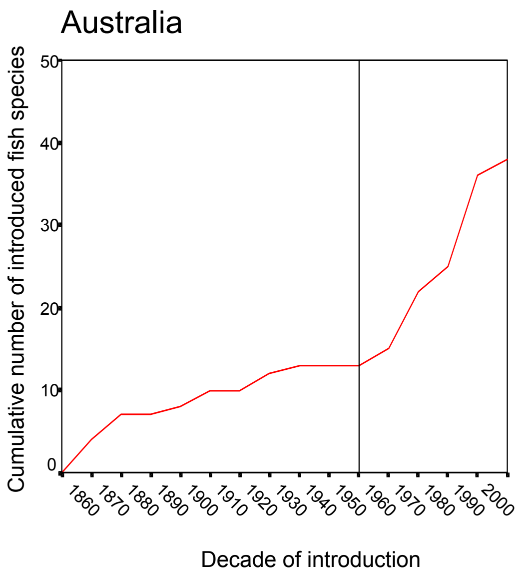


Figure B19 Cumulative number of introduced freshwater fish species in Australia

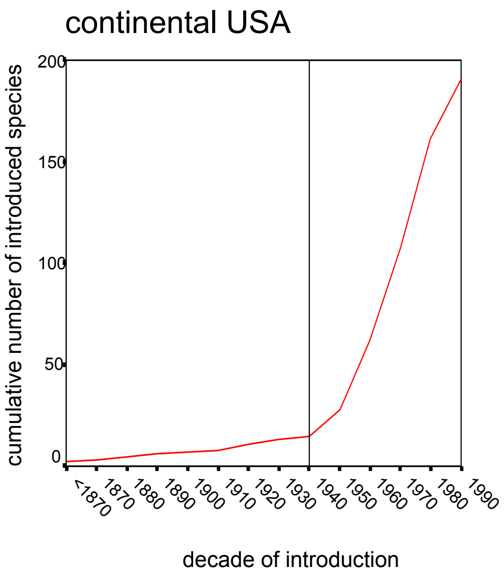


Figure B20 Cumulative number of introduced freshwater fish species in the continental United States

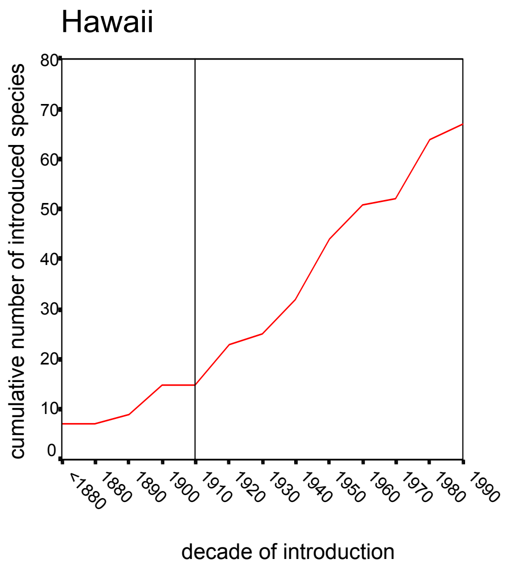


Figure B21 Cumulative number of introduced freshwater fish species in Hawaii

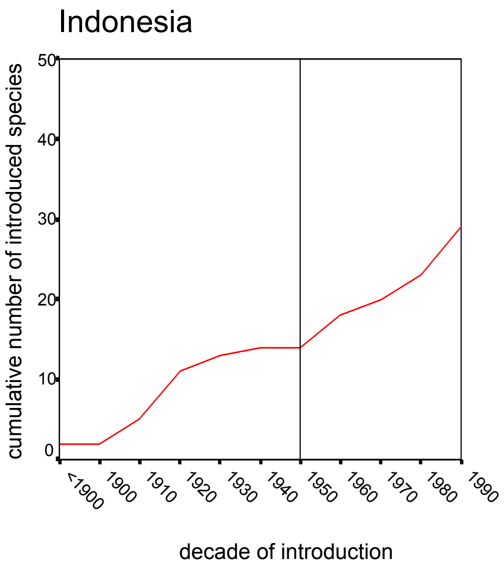


Figure B22 Cumulative number of freshwater fish species in Indonesia

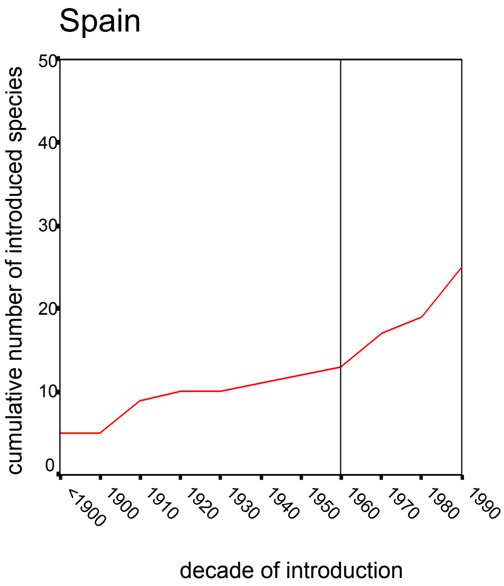


Figure B23 Cumulative number of introduced freshwater fish species in Spain

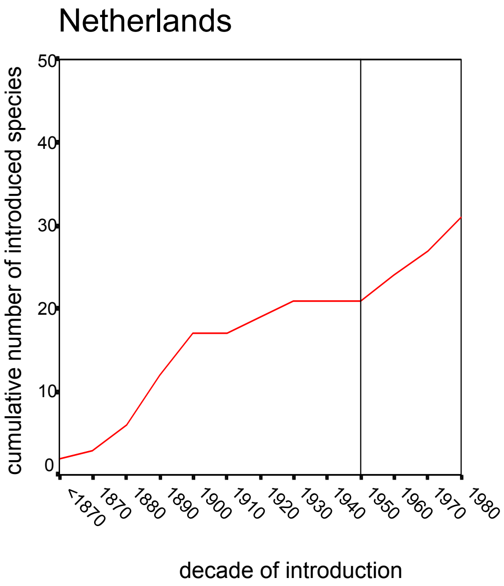


Figure B24 Cumulative number of introduced freshwater fish species in the Netherlands

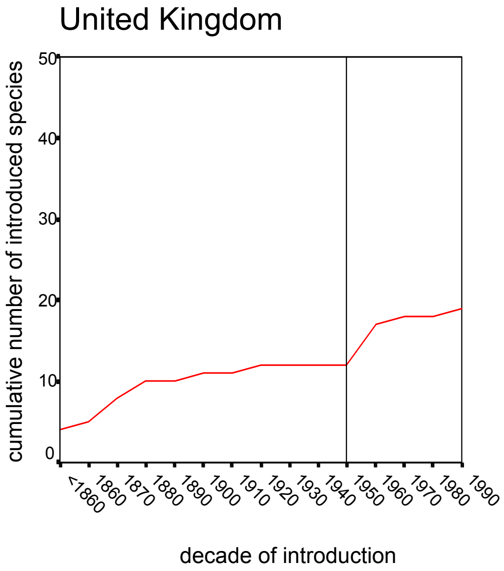


Figure B25 Cumulative number of introduced fish species in Great Britain

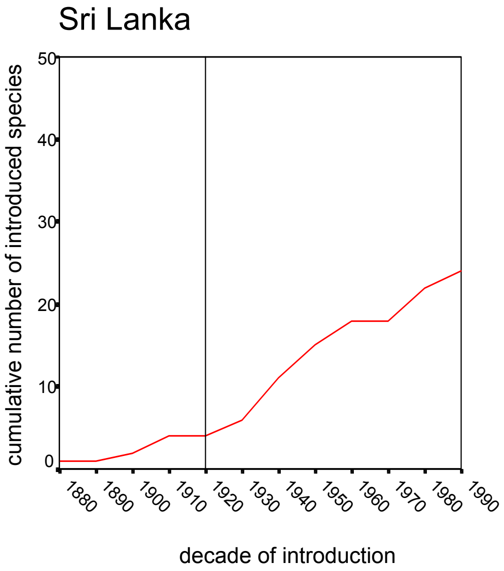


Figure B26 Cumulative number of introduced fish species in Sri Lanka

APPENDIX C

Table C1 Statistical summary for comparison of dissolved oxygen concentrations in isolated pools, vegetation barriers and main channel sites for the seine net sampling program in the Ross River catchment

	Isolated pool [1]	**Vegetation barrier [2]**	**Main channel [3]**
Sample size	15	15	15
MAX(DO2)	10.12	6.72	10.73
MIN(DO2)	0.31	0.40	5.35
median. DO2 (mg/L)	1.71	0.51	7.5
	comp 1/2	comp 2/3	comp 3/1
Mann- Whitney U	69.00	5.00	13.00
p	0.074	<0.001	<0.001

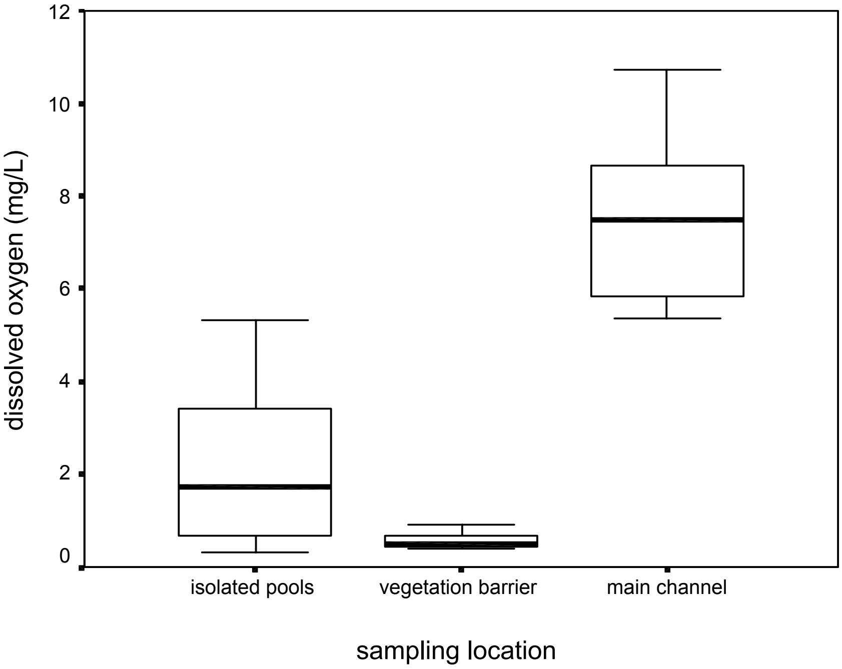


Figure C1 Comparison of dissolved oxygen concentrations in isolated pools, vegetation barriers and main channel sites in the Ross River catchment

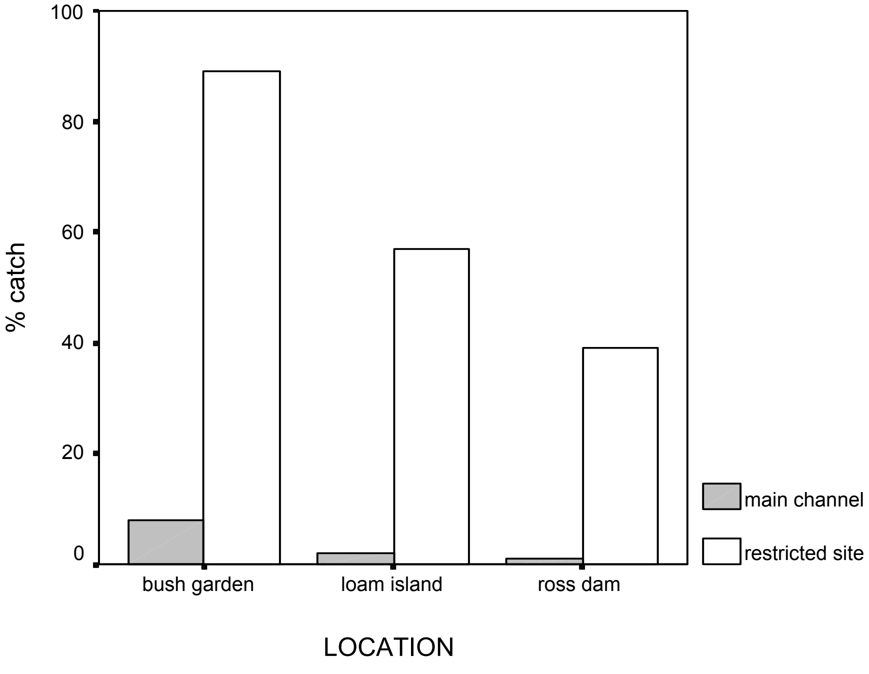


Figure C2 Percentage of non-indigenous fish in seine net samples collected in Sept. 2003 from blocked island channel, isolated wetland and adjacent main channel sites in the Ross River catchment (data collected by M. Sjoquist, S. Fagerberg, K. Konnaris, A. Hayes, R. Rodriguez and B. Hay, School of Tropical Biology, JCU)

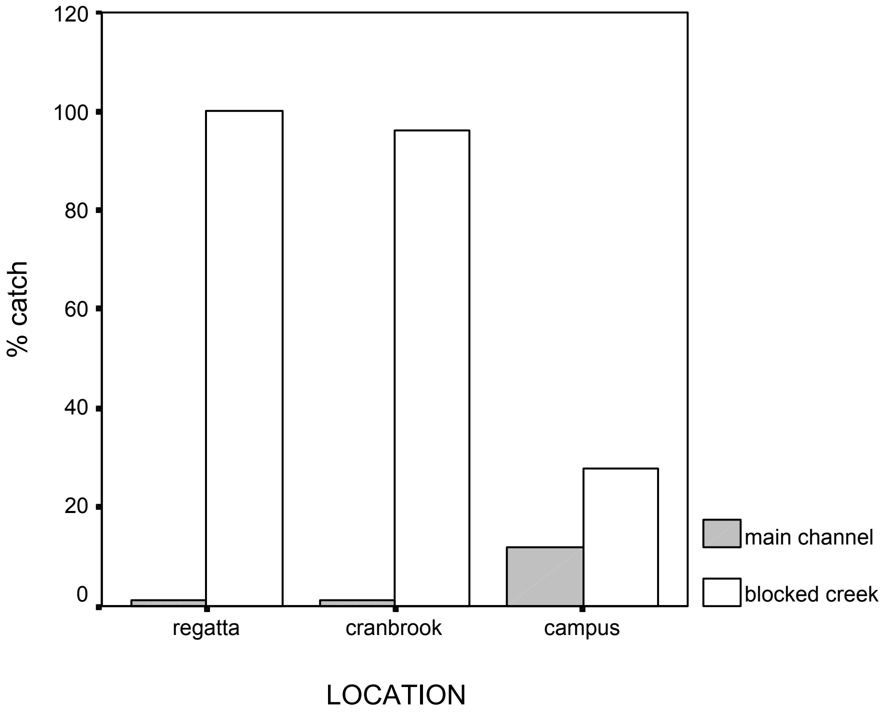


Figure C3 Percentage of non-indigenous fish in seine net samples collected in Sept. 2003 from blocked creek and adjacent main channel sites in the Ross River catchment (data collected by M. Sjoquist, S. Fagerberg, K. Konnaris, A. Hayes, R. Rodriguez and B. Hay, School of Tropical Biology, JCU)

APPENDIX D

Table D1 Morphometric data summary for the four predator species (Mouth almighty, Barramundi, Tarpon and Spangled perch), and the four prey species (Fly-specked hardyhead, Agassiz’s glassperch, Mozambique mouthbrooder and Mosquitofish), including the mean capture time of prey species, and classification of predator and prey species

Species	n	l:h	(c)		(b)	c/b	ψ	Dsp/ Nso	Fast/ Slow
Predator									
Tarpon	5	-	6.66	>	2.50	2.664	-	-	Fast
Spangled perch	10	-	2.84	>	1.67	1.701	-	-	Fast
Mouth almighty	15	-	2.10	<	2.56	-0.820	-	-	Slow
Barramundi	15	-	1.77	<	2.82	-0.628	-	-	Slow
Prey									
indigenous									
Fly-specked hardyhead		6.2	2.49	>	1.67	1.491	0.56	Nso	Fast
Agassiz’s glassperch		3.48	4.04	>	2.95	1.373	0.35	Dsp	Fast
non-indigenous									
Mozambique mouthbrooder		3.52	1.20	<	2.16	-0.555	0.16	Dsp	Slow
Mosquitofish		5.85	1.26	<	1.65	-0.764	0.13	Nso	Slow

l:h = mean body length: body height ratio; c = mean caudal fin aspect ratio; b = mean body height:width ratio; c/b = speed index; ψ = mean prey capture time per mm body length (secs); Dsp = deep-bodied and spiny-finned; Nso = Narrow-bodied and soft-finned

Table D2 Dorsal and pelvic fin spine measurements for Agassiz’s glassperch and the Mozambique mouthbrooder

Mean Dorsal and Pelvic fin spine length (DFSL) and (PFSL)								
	Glass perch				**Mozambique mouthbrooder**			
Body Depth (mm)	**DFSL (mm)**	**sd**	**PFSL (mm)**	**sd**	**DFSL (mm)**	**sd**	**PFSL (mm)**	**sd**
7	6.83	.408	4.67	.516	2.32	.464	2.86	.306
8	7.00	.000	4.50	.548	2.54	.416	3.05	.151
9	7.90	.738	5.00	.000	3.00	.000	3.14	.233
10	8.80	.632	5.60	.516	3.15	.328	3.80	.377
11	9.29	.756	6.00	.000	3.92	.640	4.15	.375
12	10.10	.594	6.60	.632	4.27	.704	4.73	.458
13	10.41	.795	6.88	.600	4.25	.337	4.67	.492
14	11.60	.516	6.90	.316	5.10	.211	5.10	.316
15	12.00	.000	7.20	.422	5.55	.522	6.00	.000
Mean Dorsal and Pelvic fine spine width (DFSW) and (PFSW)								
Body Depth (mm)	**DFSW (mm)**	**sd**	**PFSW (mm)**	**sd**	**DFSW (mm)**	**sd**	**PFSW (mm)**	**sd**
7	0.029	.0011	0.026	.0033	0.013	.0011	0.012	.0033
8	0.030	.0036	0.028	.0050	0.016	.0013	0.017	.0024
9	0.035	.0011	0.028	.0033				
10	0.040	.0028	0.035	.0023	0.018	.0009	0.021	.0018
11	0.040	.0089	0.043	.0027	0.019	.0011	0.023	.0018
12	0.049	.0030	0.044	.0026	0.023	.0011	0.024	.0009
13	0.049	.0046	0.044	.0057	0.022	.0020	0.025	.0041
14	0.052	.0009	0.044	.0038	0.023	.0011	0.026	.0033
15	0.062	.0022	0.063	.0044	0.030	.0054	0.033	.0081
16	-	-	-	-	0.032	.0022	0.036	.0057
17	-	-	-	-	0.038	.0036	0.041	.0027
18	-	-	-	-	0.038	.0033	0.042	.0040
19	-	-	-	-	0.038	.0022	0.043	.0050
20	-	-	-	-	0.037	.0041	0.046	.0063

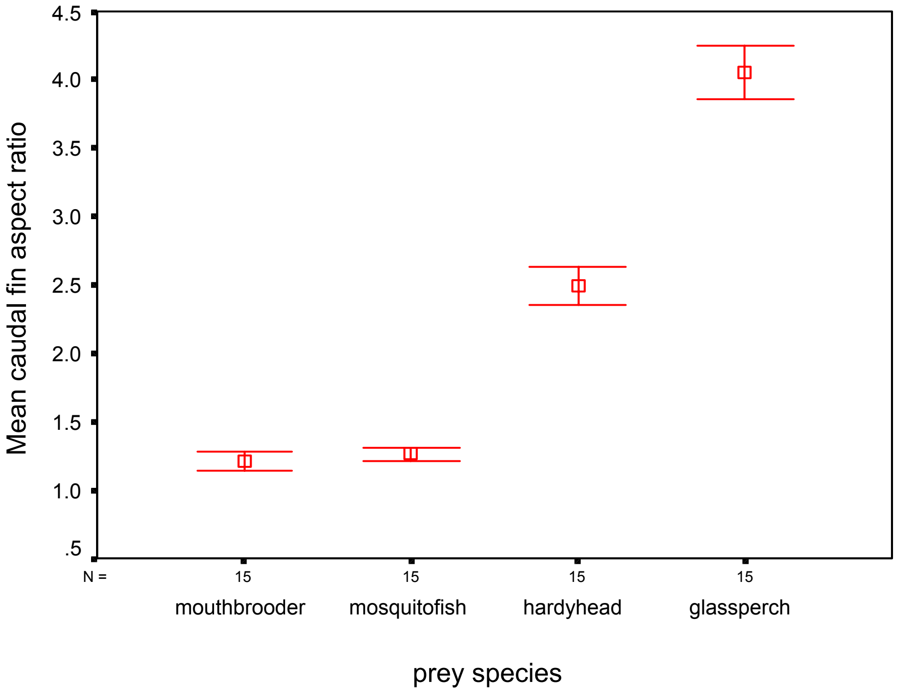


Figure D1 Mean caudal fin aspect ratio for selected prey species

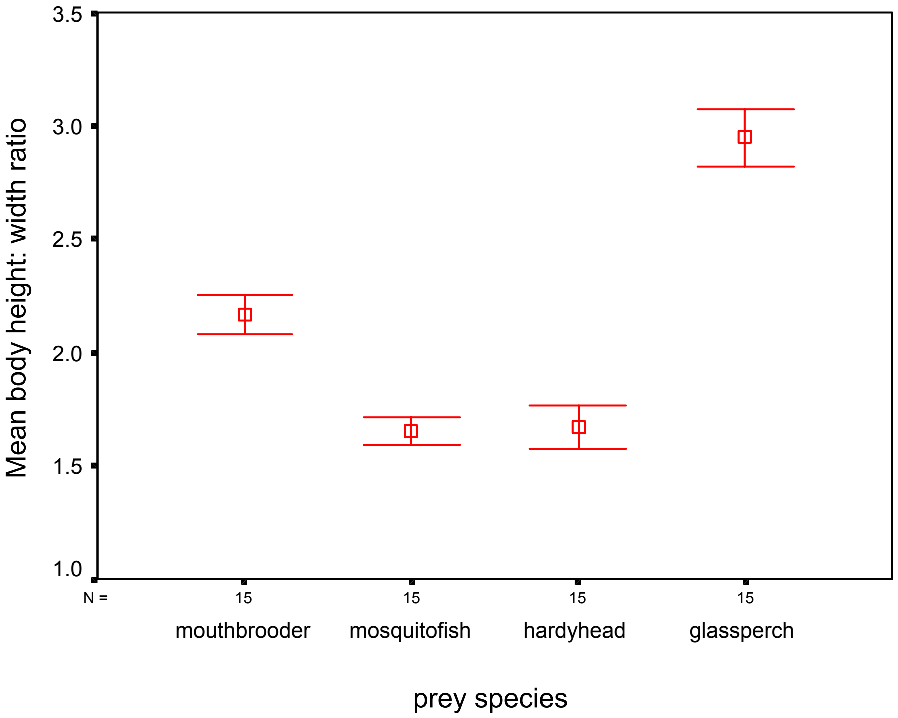


Figure D2 Mean body height : width ratio for selected prey species

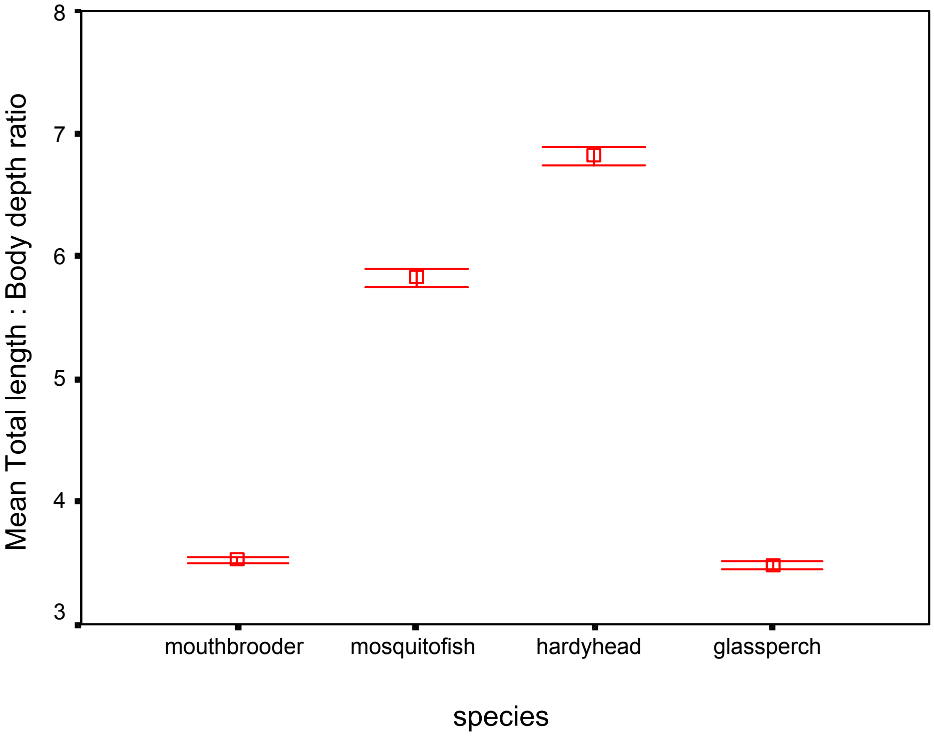
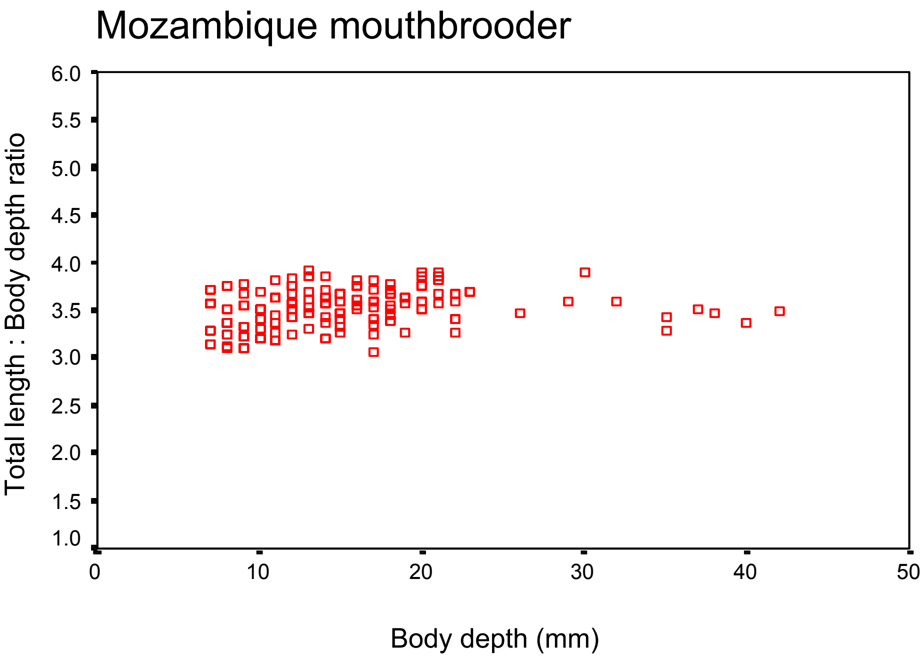
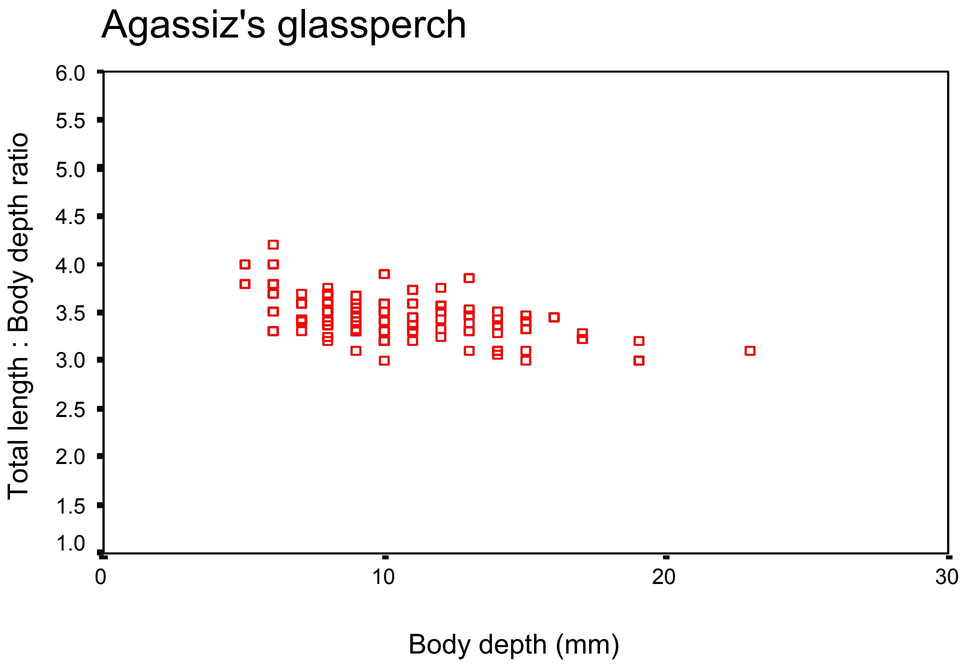


Figure D3 Mean total length : body- depth ratio for selected prey species



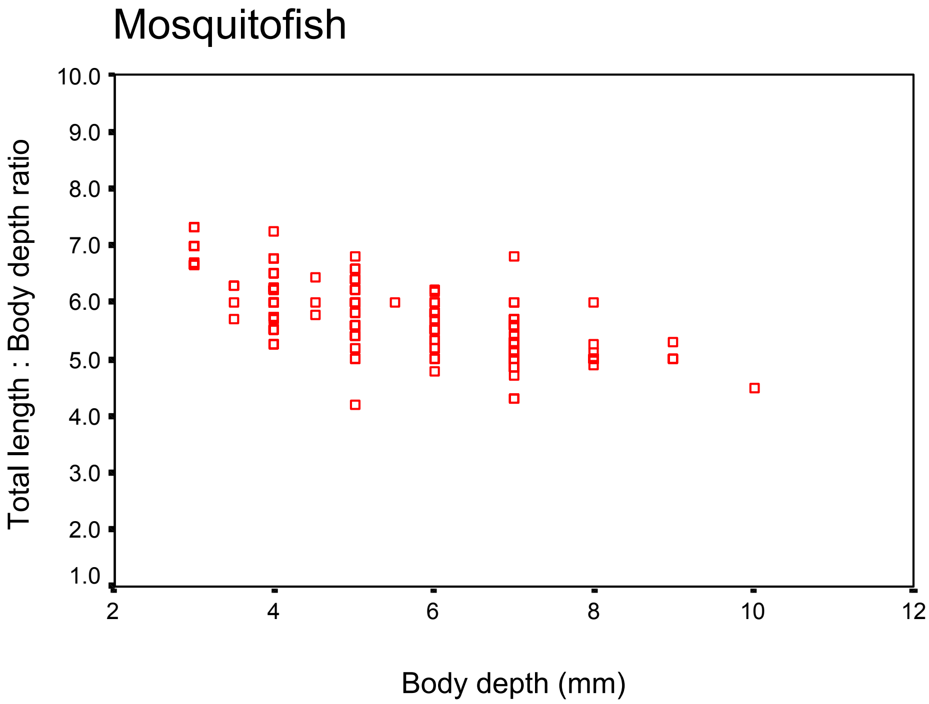
mean ratio: 3.42 (sd. 0.198; range: 3.06-3.92)

Figure D4 Total length : body-depth ratio versus body depth for Mozambique mouthbrooder juveniles



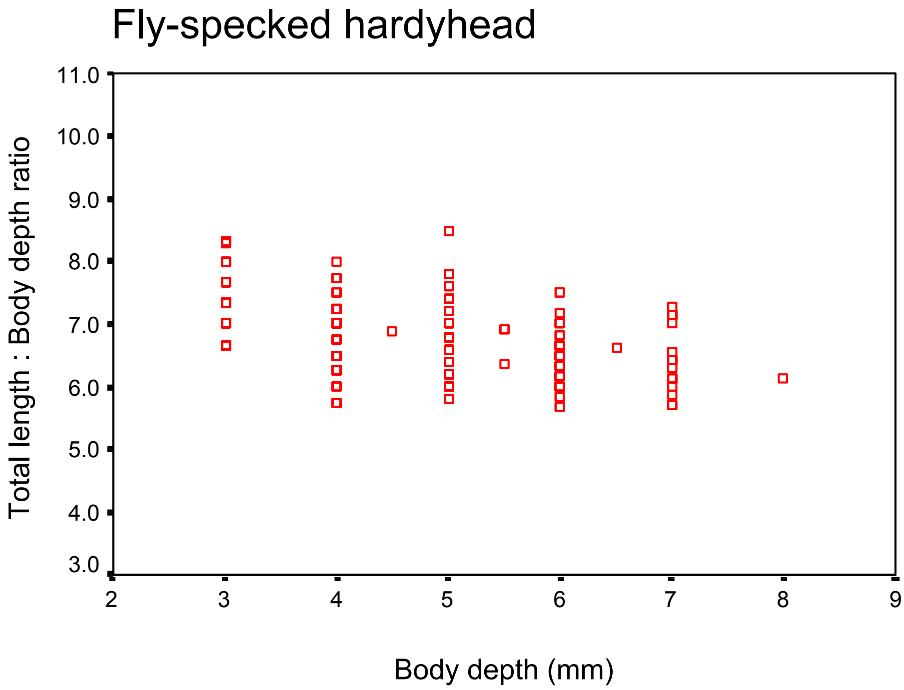
mean ratio: 3.48 (sd.0.218; range: 3.00-4.20)

Figure D5 Total length : body-depth ratio versus body depth for Agassiz's glassperch



mean ratio: 5.82 (sd. 0.597; range: 4.20-7.30)

Figure D6 Total length : body-depth ratio versus body depth for the Mosquitofish



mean ratio: 6.82 (sd. 0.642; range: 5.67-8.50)

Figure D7 Total length : body-depth ratio versus body depth for the Fly-specked hardyhead

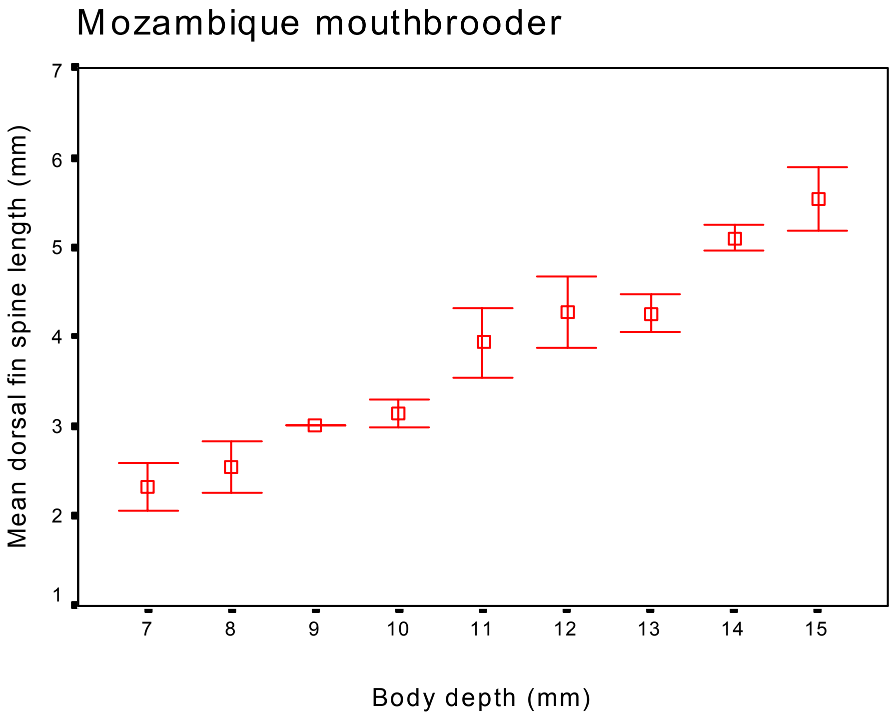


Figure D8 Mean dorsal fin spine length for different body-depth classes of Mozambique mouthbrooder

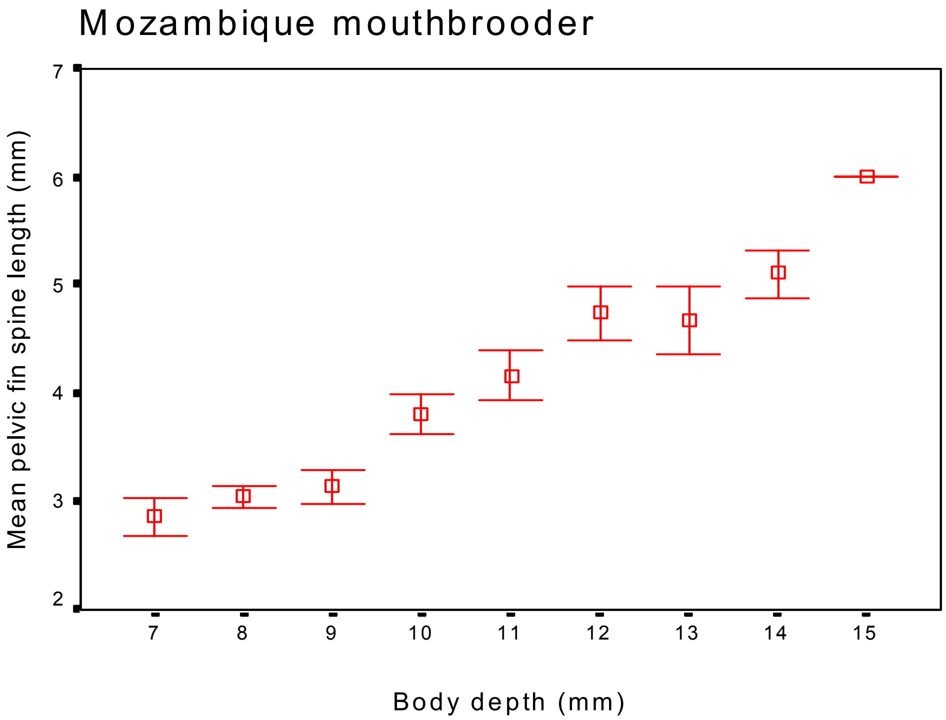


Figure D9 Mean pelvic fin spine length for different body-depth size classes of Mozambique mouthbrooder

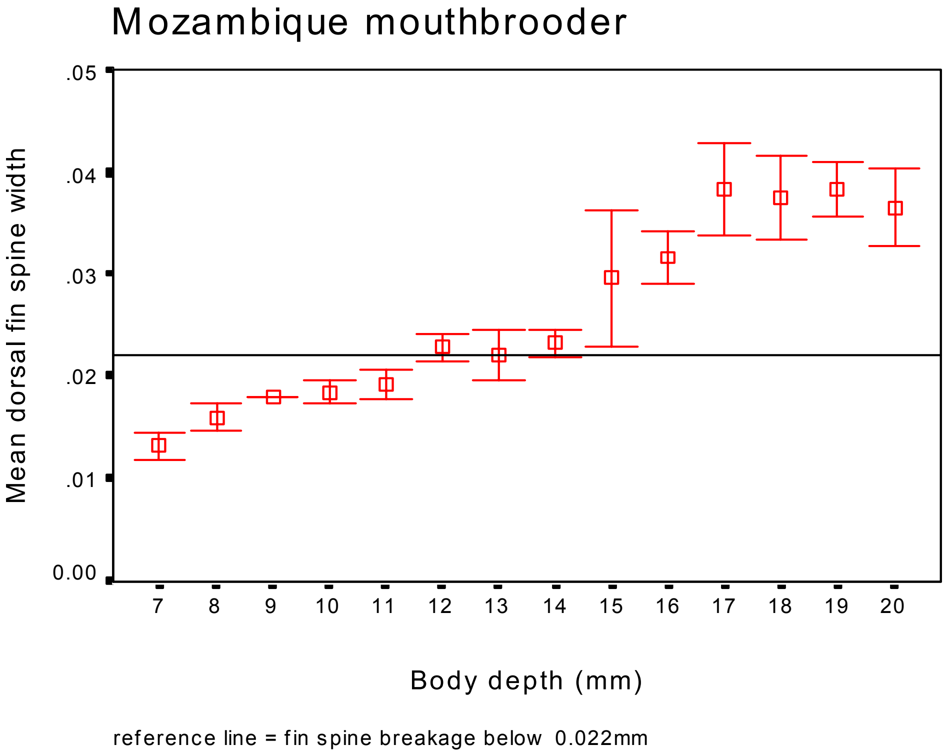
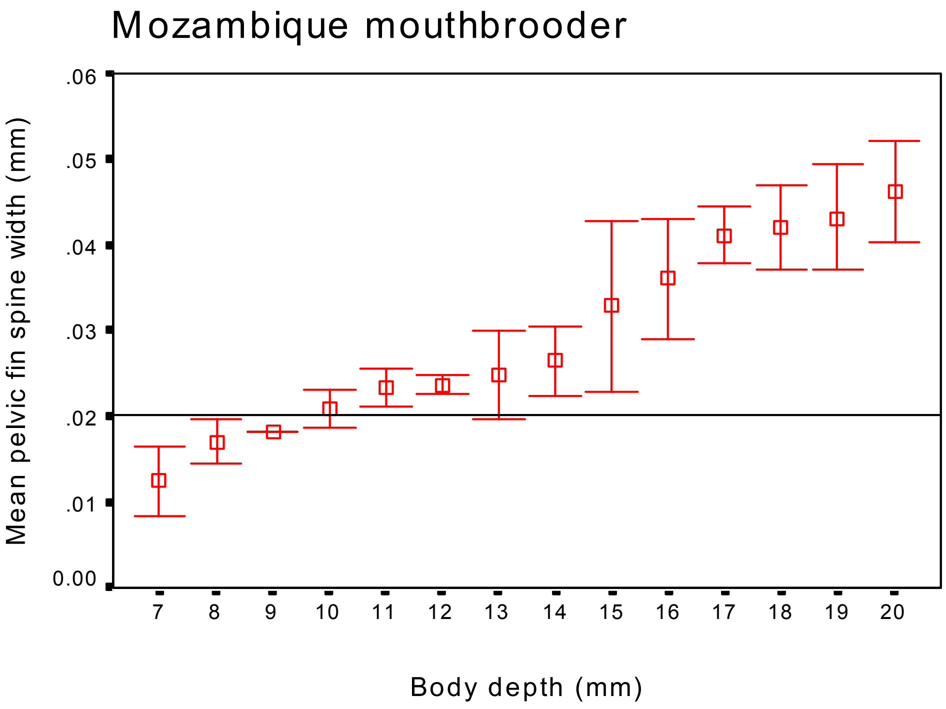


Figure D10 Mean dorsal fin spine width for different body-depth size classes of Mozambique mouthbrooder



reference line : spine breakage below 0.020mm

Figure D11 Mean pelvic fin spine width for different body-depth size classes of Mozambique mouthbrooder

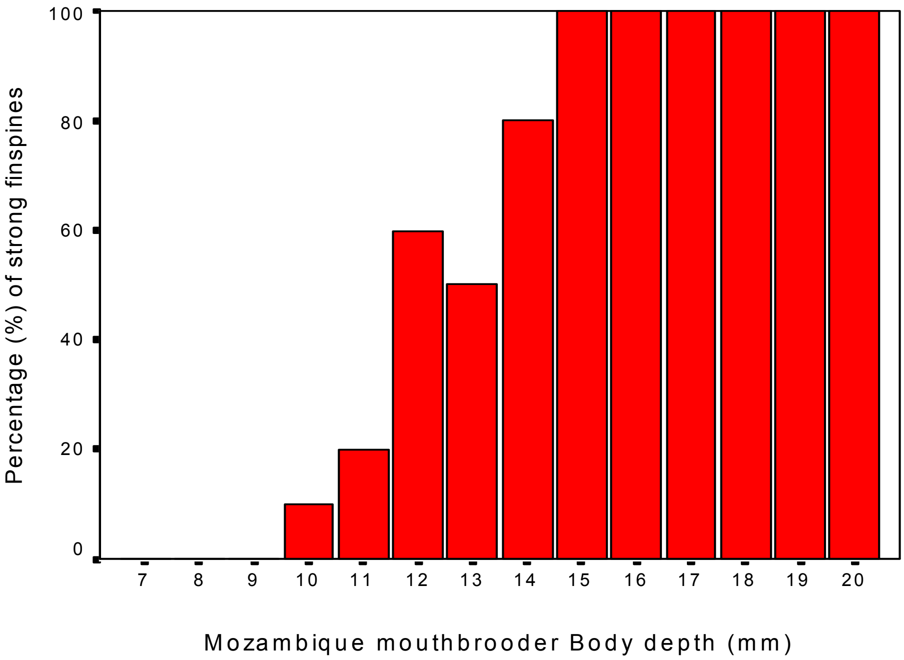


Figure D12 Percentage of fin spines above break threshold for different body-depth size classes of Mozambique mouthbrooder

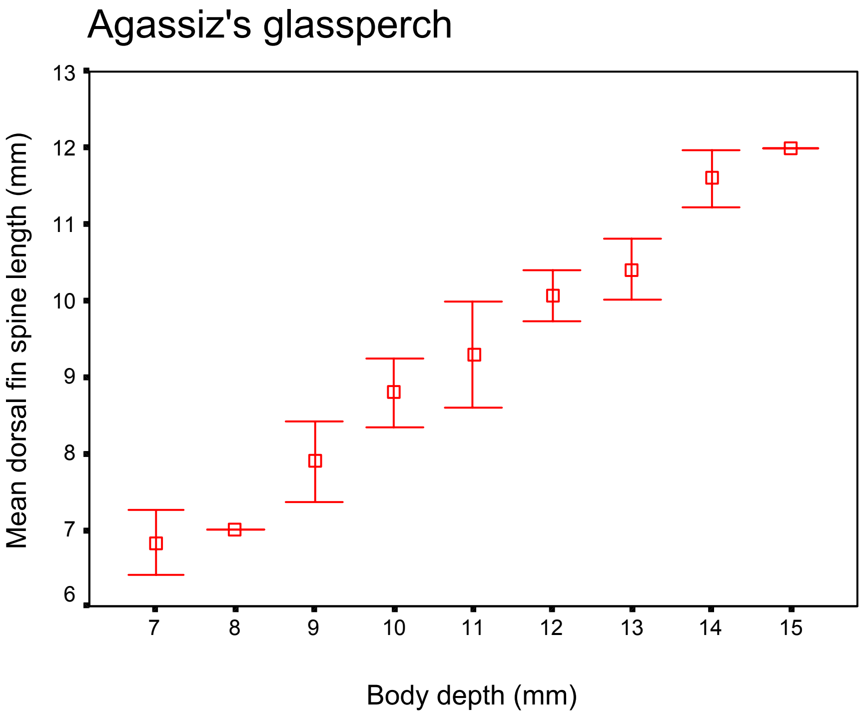


Figure D13 Mean dorsal fin spine length for different body-depth size classes of Agassiz's glassperch

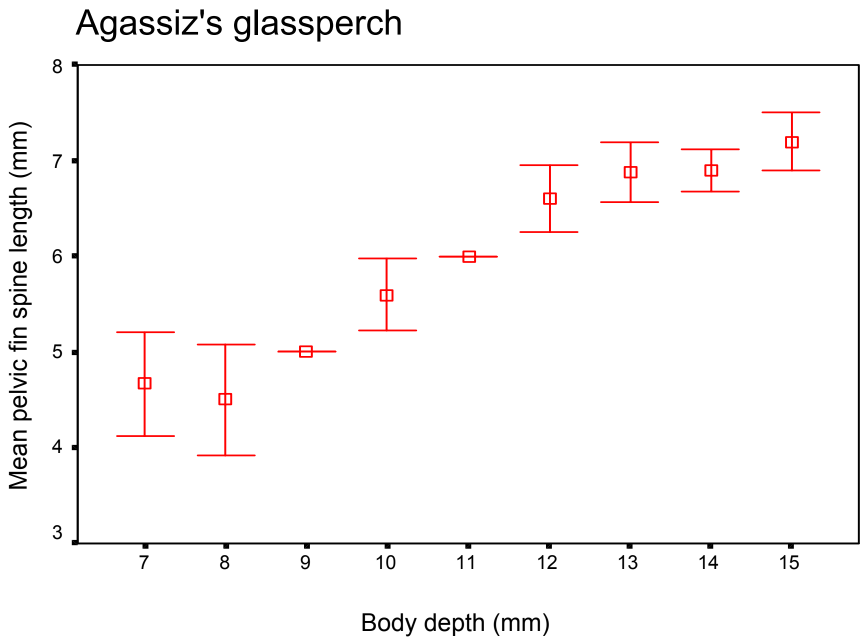


Figure D14 Mean pelvic fin spine length for different body-depth size classes of Agassiz's glassperch

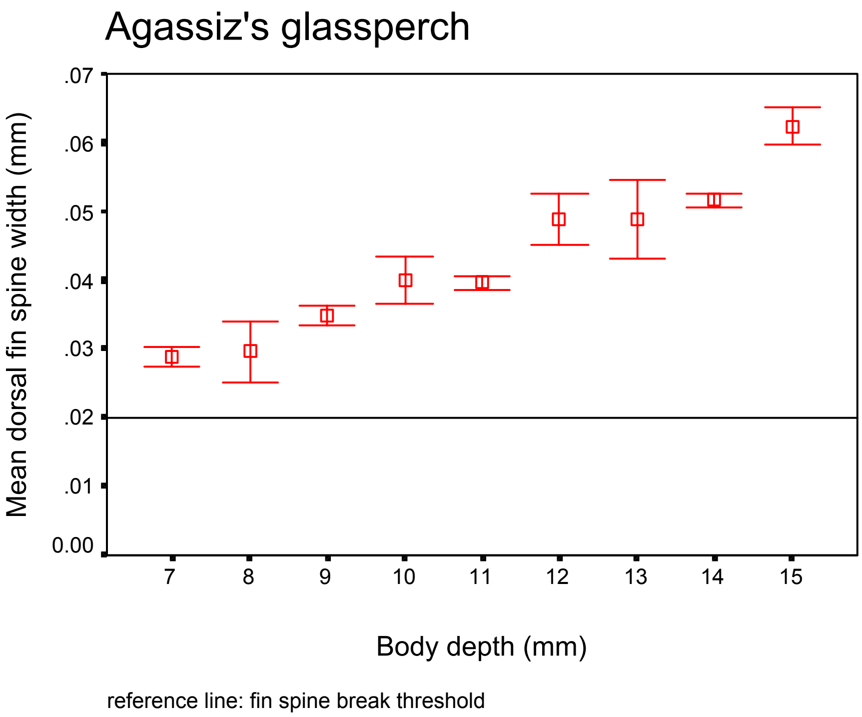


Figure D15 Mean dorsal fin spine width for different body-depth size classes of Agassiz’s glassperch

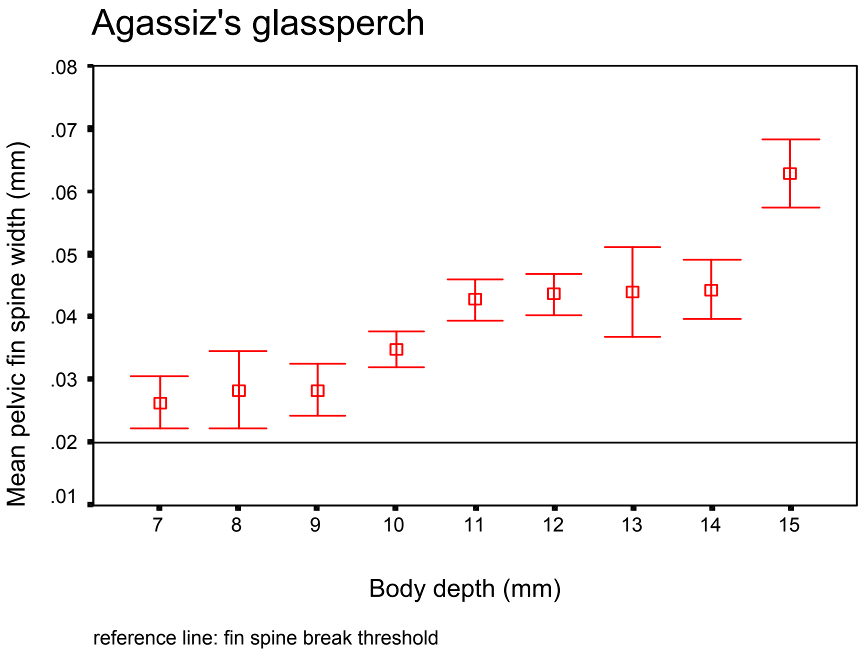


Figure D16 Mean pelvic fin spine width for different body-depth size classes of Agassiz’s glassperch

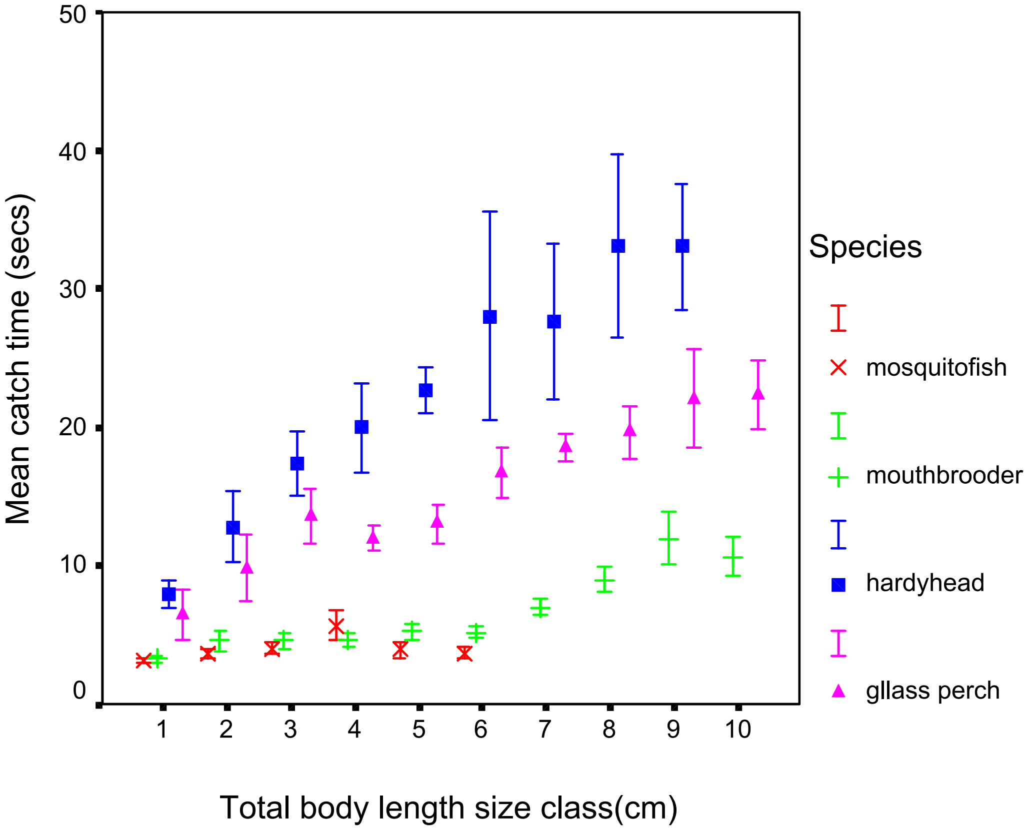


Figure D17 Change in mean catch time with body length for selected prey species

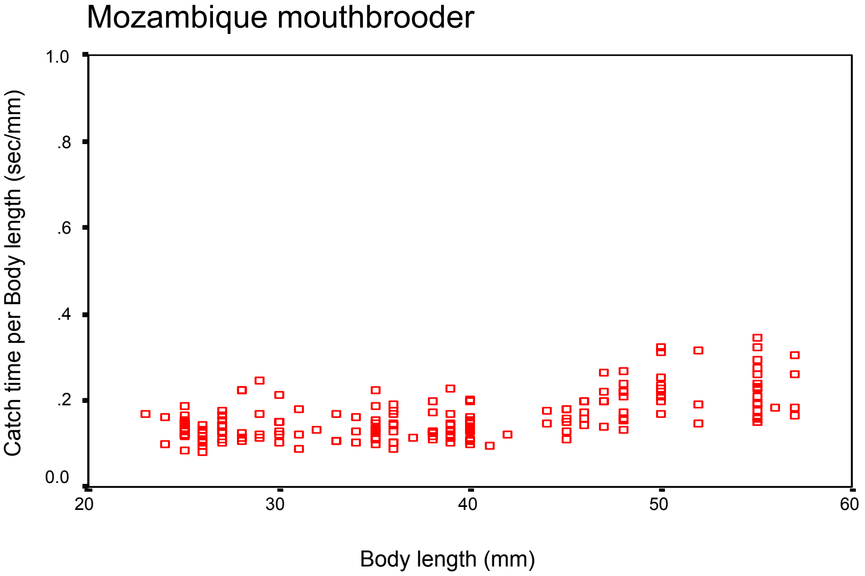


Figure D18 Catchability index versus body length for Mozambique mouthbrooder

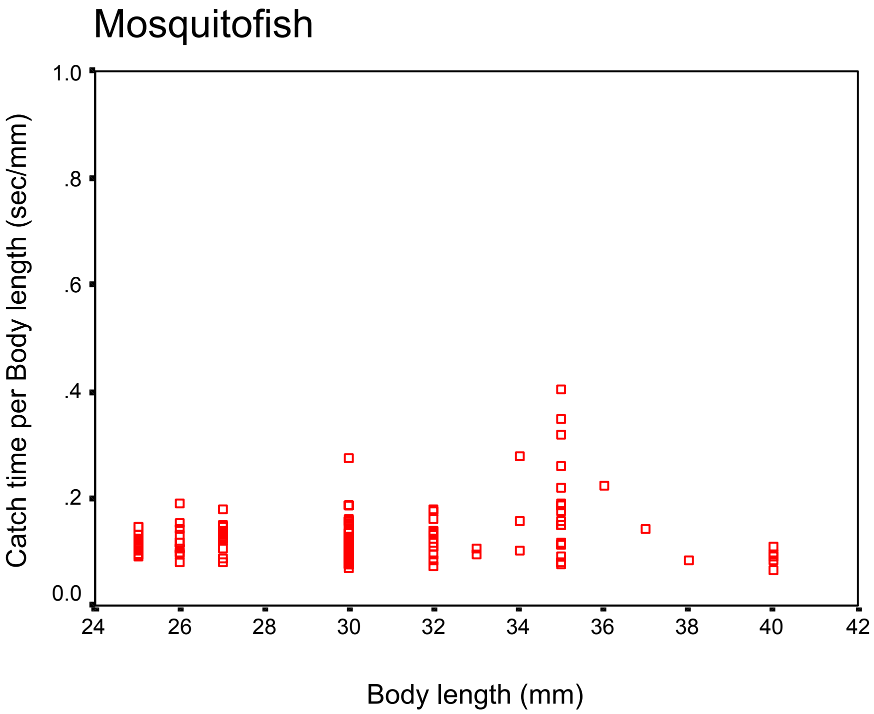


Figure D19 Catchability index versus body length for the Mosquitofish

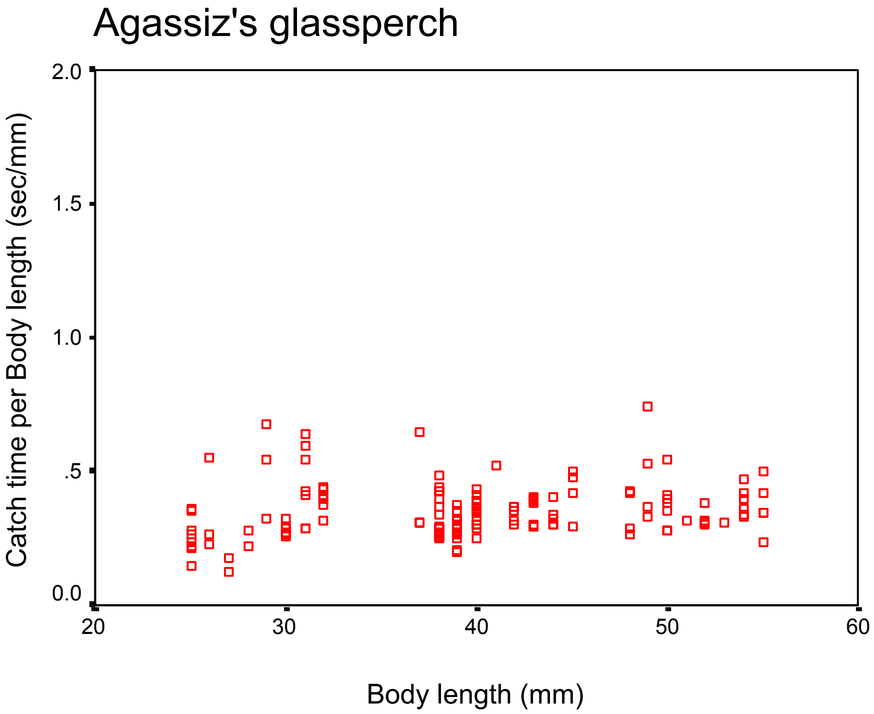


Figure D20 Catchability index versus body length for Agassiz’s glassperch

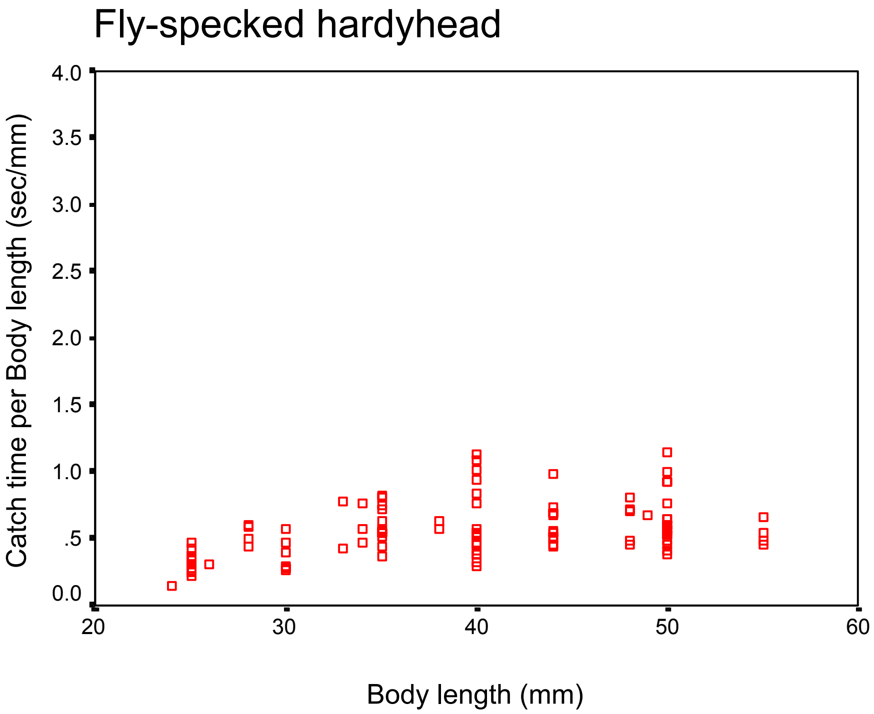


Figure D21 Catchability index versus body length for the Fly-specked hardyhead

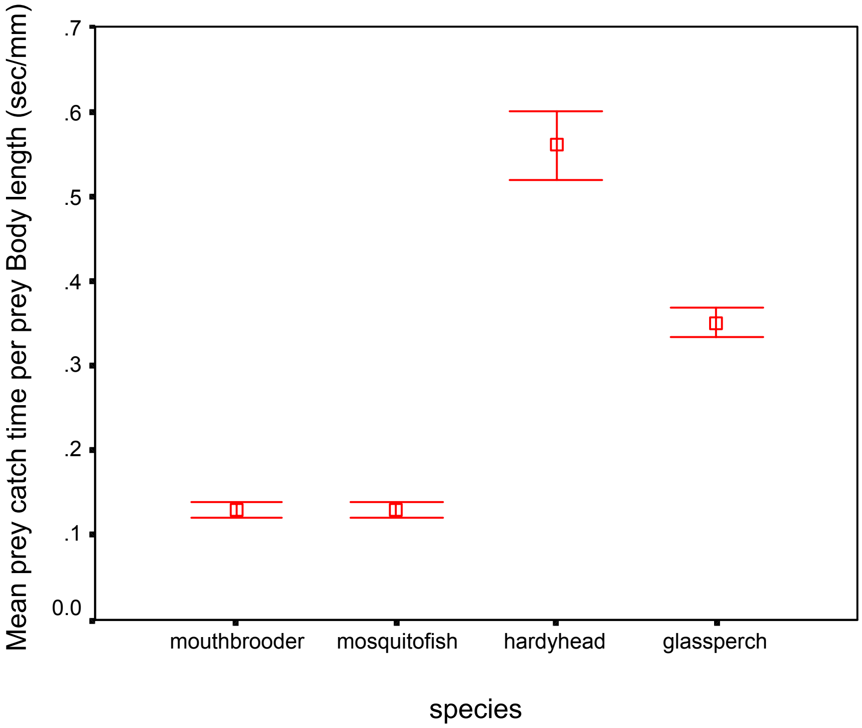


Figure D22 Mean catchability index for selected prey species

APPENDIX E

Table E1 Taxonomic reference material and other sources used for identification of fish parasites

Monogenean species on non-indigenous hosts

- *Gyrodactylus bullatarudis* (Host: Guppy, *Poecilia reticulata*) – (Turnbull 1956; Rogers and Wellborn 1965; Kritsky and Fritts 1970; Harris 1986; Dove and Ernst 1998; Richards *et al*. 2000).
- *Salsuginus heterocliti* (Host: Mosquitofish, *Gambusia holbrooki*) – (Murith and Beverley-Burton 1985; Rand and Wiles 1987).
- *Urocleidoides* sp. (Host: Platy, *Xiphophorus maculatus*) – (Mizelle and Price 1964; Hanek and Fernando 1972; Suriano 1987).
- *Cichlidogyrus sclerosus* (Host: Mozambique mouthbrooder, *Oreochromis mossambicus*) – (Paperna 1960; Parerna and Thurston 1969).

Monogenean species on indigenous hosts

- hosts: Eastern Queensland Rainbowfish and Fly-specked Hardyhead (D. Corlis, unpubl. data – PhD thesis, Department of Zoology, JCUNQ, Townsville, Australia);

- hosts: Agassiz's glassperch, Empire gudgeon, Fire-tailed gudgeon, Purple-spotted gudgeon and Mouth almighty (Fletcher 1996, Hons thesis, Department of Parasitology, UQ)

Other parasites

- Nematodes: (Hartwich 1974; Chabaud 1975a,b; Anderson and Bain 1982; Anderson 1992).
- Cestodes: (Khalil *et al.* 1994).
    - *Bothriocepalus aechilognathi* (Yamaguti 1934; Pool and Chub 1985; Schultz and Di Cave 1992; Dove *et al.* 1997; A. Dove, *pers. comm.*)
- Digeneans: (D. Barton, Department of Zoology, JCUNQ, Townsville, Australia, *pers. comm.*)
- Acanthocephalans: *Telosentis* sp. nov. (Amin 1985).
- Copepods: (Host: *Tilapia mariae*) (Kabata 1992)
- Branchiura: *Argulus* sp. nov. (Host: *Oreochromis mossambicus*) (W. Poly, Department of Ichthyology, California Academy of Sciences, San Fransisco, USA, *pers. comm.*).

Table E2 Presence/absence input matrix for NMDS ordination analysis of parasite communities of indigenous and non-indigenous freshwater fish species in northern Queensland

1 present; 0 absent

Parasite species	*Host species*									
	MM	GH	PR	XM	RF	GP	HH	MA	PSG	EG
Cichlidogyrus sclerosus	1	0	0	0	0	0	0	0	0	0
Telosentis sp. 1	1	1	1	0	0	0	0	0	0	0
Argulus sp. 1	1	0	0	0	0	0	0	0	0	0
Ichthyophthirius multifiliis	1	0	0	1	0	0	0	0	0	0
Clinostomum sp.*(complanatum)*	1	1	1	0	1	1	0	1	1	0
Metacercarial cyst sp. 2	1	1	0	0	1	1	1	1	0	0
Metacercarial cyst sp.3	1	1	1	0	0	1	0	0	0	0
Echinostome sp. 1	1	1	0	0	1	1	1	1	0	0
Bothriocephalus acheilognathi	1	0	1	0	0	0	0	0	0	0
Eustrongylid sp. 2	1	0	0	0	0	0	0	0	0	0
Gyrodactylus bullatarudus	0	0	1	0	0	0	0	0	0	0
Dracunculid (Spirurida) sp. 1	0	1	0	0	1	0	0	0	0	0
Contracecum sp. 1	0	1	1	0	1	0	1	1	0	0
Contracecum sp. 2	0	0	0	0	0	0	0	0	1	0
Amphilinidea sp. 1	0	0	1	0	0	0	0	0	0	0
Salsuginus heterocliti	0	1	0	0	0	0	0	0	0	0
Urocleidoides sp. 1	0	0	0	1	0	0	0	0	0	0
Philometriodes sp. 1	0	0	0	1	0	0	0	0	0	0
Longidigitis iliocirrus	0	0	0	0	1	0	0	0	0	0
L. aunipontiformis	0	0	0	0	1	0	0	0	0	0
L. chunkyanchor	0	0	0	0	1	0	0	0	0	0
Helicirrus splendida	0	0	0	0	1	0	0	0	0	0
H. megaanchor	0	0	0	0	1	0	0	0	0	0
Metacercarial cyst sp. 1	0	0	0	0	1	1	0	1	0	0
Taenid sp. 1	0	0	0	0	1	0	0	0	1	0
Myxozoan sp. 2 (cf. *Hennegrya*)	0	0	0	0	1	0	0	0	0	0
Philometrid sp. 1	0	0	0	0	1	0	0	0	0	0
Longidigitis curvatus	0	0	0	0	0	0	1	0	0	0
Recurvatus chelatus	0	0	0	0	0	0	1	0	0	0
Pseudodactylogyroides kalingaensis	0	0	0	0	0	0	0	0	1	1
Metacercarial cyst sp. 4	0	0	0	0	0	1	0	0	0	0
Metacercarial cyst sp. 5	0	0	0	0	0	0	0	0	1	0
Spirocamallanus sp. 1	0	0	0	0	0	0	0	0	0	0
Haplocleidus sp. 1	0	0	0	0	0	1	0	0	0	0
Eustrongyloides sp. cf *excisus*	0	0	0	0	0	1	0	0	0	0
Pseudodactylogyroides kholoensis	0	0	0	0	0	0	0	1	0	0
Ergasilus intermedius	0	0	0	0	0	0	0	1	0	0
Taenid sp. 2	0	0	0	0	0	0	0	1	0	0
Philometrid sp. 1	0	0	0	0	0	0	0	1	0	0
Spirocamellanus sp. 1	0	0	0	0	0	0	0	1	0	0
Spinitectus sp. 1	0	0	0	0	0	0	0	1	0	0

MM = Mozambique mouthbrooder; GH = Mosquitofish; PR = Guppy; XM = Platy; RF = Eastern Queensland rainbowfish; GP = Agassiz's glassperch; HH = Fly-specked hardyhead; MA = Mouth almighty; PSG = Purple-spotted gudgeon; EG = Empire gudgeon

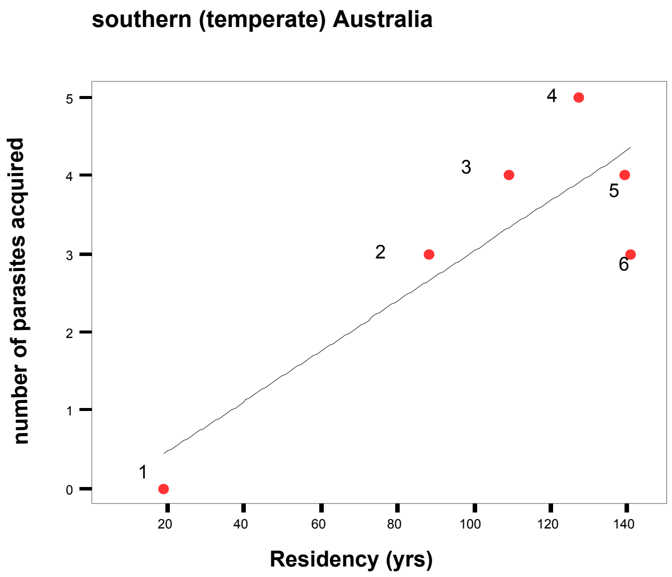


Figure E1 Parasites acquired by non-indigenous freshwater fishes in relation to residency time in southern Australia (Host: 1 = Weather loach; 2 = Mosquitofish; 3 = Rainbow trout; 4 = Goldfish; 5 = Brown trout; 6 = Redfin perch

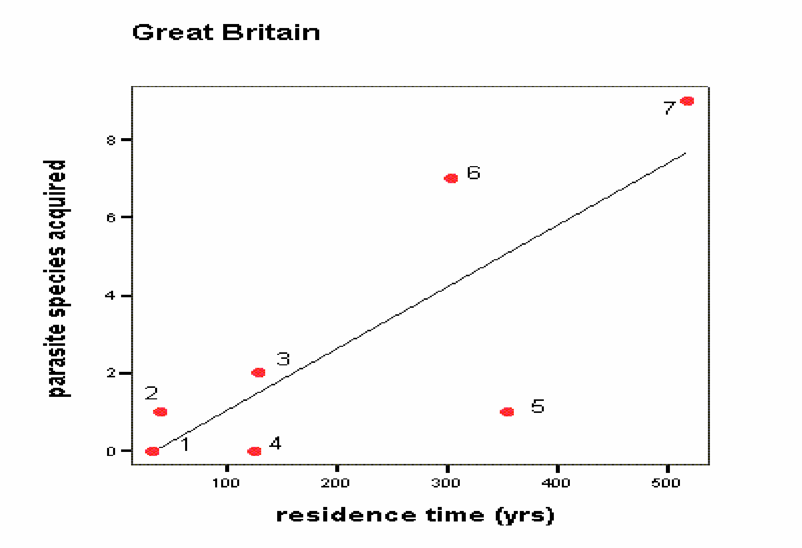


Figure E2 Parasites acquired by non-indigenous freshwater fishes in relation to residency time in Great Britain (1 = Grass carp; 2 = Pike-perch; 3 = Orfe; 4 = Largemouth bass; 5 = Goldfish; 6 = Crucian carp; 7 = European carp