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Review of the impacts of introduced aquarium fish species that have established wild populations in Australia

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1. Introduction

1.1 Reason for this review

The introduction and spread of exotic species in various parts of the world is regarded by many as a major threat to global biodiversity (Vitousek et al. 1997; Sakai 2001; Kolar and Lodge 2001; Lee 2002; Dudgean et al. 2006), and this threat applies substantially to freshwater fishes (Courtenay 1990; Courtenay and Stauffer, 1990; Courtenay and Moyle, 1992: Fuller et al. 1999; Canonico et al. 2005). There are many instances where introductions of exotic species ranging from micro-organisms to vertebrates have had unexpected consequences for the native fauna and flora in both terrestrial and aquatic ecosystems (IUCN 2001; Global Invasive Species Programme webpage). When reviewing global causes of species decline, Reid et al. (2005) noted that the introduction of non-native, exotic species is the major cause of extinctions. This is especially so in freshwater ecosystems such as lakes (Sala et al. 2000).

In Australia, the introductions of species such as the cane toad, prickly pear, foxes, rabbits, and rodents are among the higher profile biological invasions (Low 2001), although many Australians are also now aware of potential threats posed by large, introduced freshwater fish such as common carp (Roberts and Tilzey 1997). It is less likely, however, that Australians are generally aware of the potential ecological impacts of other introduced freshwater fish species, especially the small fish species prevalent in the freshwater aquarium trade.

Small fish species can be just as great a threat to native fish species as large fish species. For example, *Gambusia holbrooki* (mosquitofish) was introduced to Australia for the control of mosquito larvae. Although this species is relatively small (maximum size < 6 cm), *Gambusia* has now been linked to ecological impacts on the native freshwater fauna in most of the countries to which it has been introduced throughout the world (IUCN 2001). The potential impact of other small fish that are introduced into the wild may also be significant and small fish should not be discounted because of their size or the fact that they are ornamental and relatively benign in an aquarium.

Many ornamental fish are brought into Australia for the aquarium trade and between 12 and 14% of Australians are thought to keep aquaria (McNee, undated). These fish generally have no obvious value arising from their being released into Australia's waterways. Despite this, there has been a steady increase in the number of exotic freshwater ornamental fish species that have become established in Australian waterways over the past 20-30 years (Arthington et al. 1999; Lintermans 2004; Koehn and Mackenzie 2004). It is noteworthy that ornamental fish species account for a majority of the recent fish introductions to Australian freshwaters (Fig. 1.1) and constitute a 'new wave' of fish introductions that far exceeds that which occurred in the late 1800's with the influx of European immigrants.

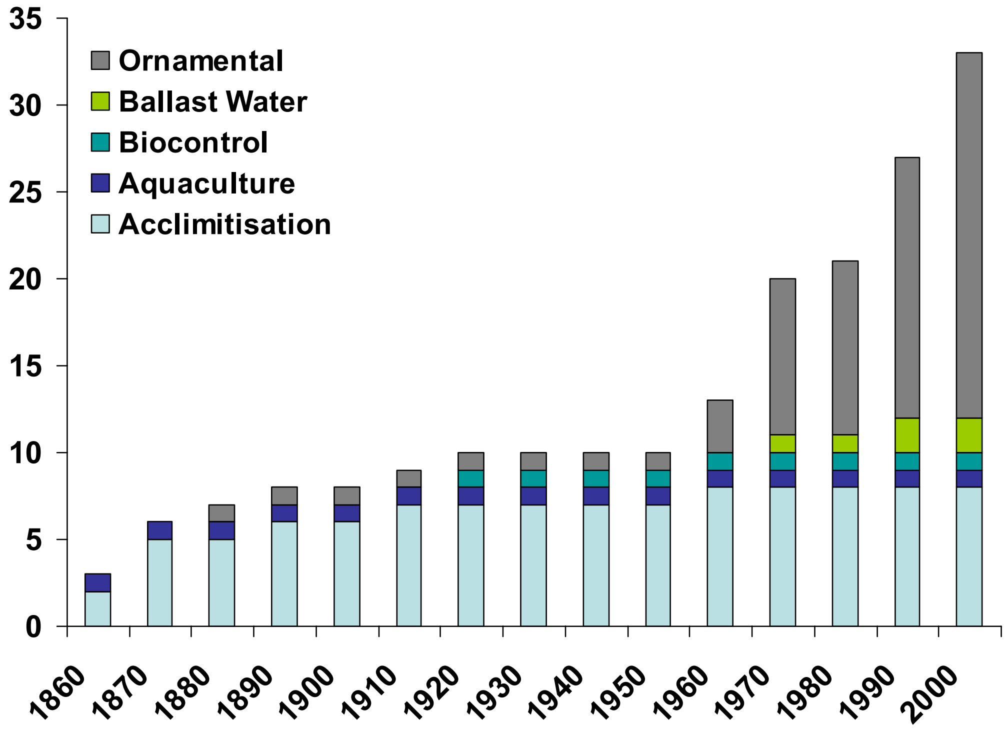


Figure 1.1: The rate of introduction of exotic fish species into Australian freshwaters categorised by sectors responsible for importation (courtesy Dr T. Peacock, Invasive Animals CRC as derived from Lintermans (2004)).

A recent study conducted by Casal et al. (1999) documented the status of exotic freshwater fish in Oceania and found that although Australia had the highest diversity of freshwater fish in the Oceania region, the proportion of exotic species that were established in the wild (10%) was among the lowest of the countries considered. However, Casal et al. (1999) stated that 11 of the most commonly introduced species in Oceania were considered to have adverse effects in at least one country and five of these were ornamental fish species (i.e., *Oreochromis mossambicus*, *Tilapia zillii*, *Carassius auratus, Xiphophorus hellerii* and *Poecilia reticulata*). The impacts of such ornamental fish species in Australia have received far less attention than those of common carp, but already a number of species (e.g., goldfish, tilapia, oriental weatherloach and a few poeciliids/platys) have been associated with some impact in some locations (Lintermans et al. 1990; Lintermans 1993; Arthington 1986, 1989, 1991; Arthington and Mitchell 1986; Arthington and Bluhdorn 1994; Arthington and Cadwallader 1996).

Risk assessment and management frameworks have been developed in Australia for ornamental fish (e.g., Arthington et al. 1999; Kailola 2000; Bomford and Glover 2004; DAFF 2005) and are used mainly for assessing the risk of importing a particular ornamental species. The underlying principles of such risk assessments are invasion theory, particularly views espoused by Moyle and Light (1996a,b). Their theory is underpinned by the biological properties of different fish species in relation to both their potential invasiveness and the nature of the receiving environment. However, not all scientists specialising in invasive fish ecology support the use of all these attributes. Invasion ecology is an inexact science and there are many uncertainties in it as well as different ways of assessing risk, none of them perfect. For example, the risk assessment framework developed by McDowall (2005) for New Zealand was preceded by a critical review of the attributes associated with invasiveness listed by Moyle and Light (1996a). McDowall (2005) only incorporated a subset of these attributes into his risk assessment framework. Of these, physiological temperature tolerances of the species were considered the most reliable criterion for evaluating the likely success of introductions in New Zealand.

The risk of a species becoming established in the wild is also related to ‘propagule pressure’ and to the number of pathways by which a species can be spread to the wild (Kolar & Lodge 2000; Lodge 2001; Ricciardi & MacIsaac 2001). This is particularly relevant for ornamental fish in the sense that the most popular species for aquaria can be expected to be as widely and densely distributed as human residences, with each aquarium constituting a potential source of propagules for establishment of these species in the wild. However, apart from the match between habitats and species tolerances, establishment in the wild will also depend on the number of pathways by which such fish are transferred from aquaria to natural waters. The latter is clearly a key process in the invasion of ornamental fish and many of the various pathways by which exotic fish are released into natural waters have been well described by Lintermans (2004). Propagule pressure and dispersal pathways are therefore key components of risk assessments relating to the invasive potential of exotic fish populations. Both need to be considered alongside species-specific impacts to determine the potential for a species to become a pest. In this sense a pest fish is defined as one which impacts on native fauna and habitats in a wide range of situations and which also has the potential to become widely established. Ideally risk assessments applied to fish predict their potential to cause adverse impacts as well as to spread widely because such attributes are not correlated for fish species as readily as they are for exotic plant, insect or mammal species that invade terrestrial environments.

Applying such risk assessment frameworks to exotic fish species for which there is a paucity of data can result in erroneous findings or interpretations and lead to a precautionary approach which may be unnecessary. On the other hand, available information often does not allow the application of rigorous protocols that provide sufficiently secure protection from adverse impacts. While a precautionary approach to managing invasive species is seen by many as wise or even essential, the aquarium industry is a key stakeholder that could be negatively affected by adverse publicity surrounding the perceived impacts of ornamental species introduced and/or established in Australian waterways. If the aquarium industry is to contribute in a meaningful way to ensuring effective management of established ornamental fish in Australia, it is likely to require more robust data on the impacts of these species than that provided by risk assessments. Furthermore, Australian environmental legislation leans heavily towards the protection of biodiversity (e.g., the EPBC Act 1999) and the application of this legislation will require much better scientific information on the effects of ornamental fish species on native biodiversity. Apart from the need for more robust data on the potential environmental impacts of aquarium fish, the aquarium industry has economic and social values and any environmental consequences of introduced species need to be considered within this context.

In summary, the issue of the impacts of established ornamental fish in Australia's waterways combines ecological, social, economic and legislative elements and there may be major knowledge gaps in terms of potential or actual impacts that need to be filled before effective management can be determined and/or the support from key stakeholder groups obtained. It is on this basis that the Department of the Environment & Water Resources (DEW) commissioned a study to review the current status and potential ecological threats posed by the freshwater ornamental fish species that have already established breeding populations in Australian waters. The purpose of this review is to identify key gaps in knowledge so that the DEW and the aquarium trade can develop a joint approach to managing the environmental consequences of the introduced species.

Although an assessment of gaps in knowledge of the environmental impacts posed by ornamental fish is required, a key component of this review is a socio-economic appraisal of the use of these fish and of the gaps in knowledge of their economic and social costs and values. This is a novel, but necessary, component of impact assessment, and is designed to assist managers of aquatic resources to better understand the issues. This will result in the aquarium industry obtaining a clearer picture of the extent of this management issue and their potential role in addressing it.

To maintain a balanced and wide-ranging approach to the issue of whether ornamental or aquarium fish pose a threat to the native Australian freshwater fauna, a number of experts in the various fields have helped prepare this report. For example, the economic chapter was prepared by Anya Richards and Charles Jubb for Meyrick & Associates, a firm of economists with specialist expertise in the field of assessing the economic and social costs of environmental problems. An expert on fish diseases (Dr Ben Diggles) prepared the chapter covering the potential threats from the spread of diseases and pathogens, while Andrew Moore, who has researched exotic fish, especially the impacts of *Gambusia* in Australia, mapped the species distributions and considered the genetic implications of the introduction of aquarium fish to Australia. Drs Jamie Corfield, Bob McDowall and Dave Rowe, all with the National Institute for Water and Atmospheric Research Ltd (Australia and New Zealand) are fish biologists with collective expertise in exotic fish and prepared the remainder of the report.

Another key factor helping to maintain a balanced approach during this study was the establishment of a review panel comprising four people, representing key stakeholders and including the aquarium trade, the scientific community and conservation interests. Such a review panel was established and charged with reviewing the report and ensuring that it presents an unbiased viewpoint.

Personnel involved are:

- Professor Angela Arthington is a senior academic with Griffith University and has specialised in assessing the ecological impacts of introduced fish into Australia over many years.

- Professor Bob Lester from the University of Queensland is also a senior academic and he has specialist knowledge of fish parasites and disease issues.

- Dr Andreas Glanznig is a senior policy analyst with the World Wildlife Fund (Australia) and has specialist knowledge of the effects of introduced species (including freshwater fish) on the conservation of Australian ecosystems.

- Jared Patrick is the owner and manager of Bay Fish (www.bayfish.com.au) a wholesale distributor of aquarium fish and he is a senior representative of the Aquarium Industry in Australia.

1.2 Scope of the review

By the late 1980s, over 1,000 fish species had been brought into Australia since European settlement (McKay 1989). Similarly, McNee (no date) indicated that 1,181 exotic species of mainly freshwater fish had been recorded in Australia over the past

40 years and only 481 of these were on the current permitted import list. More exotic freshwater fish, including non-aquarium species, will have been introduced since these reports, however, this report deals only with 'ornamental' species involved in the freshwater aquarium industry.

Currently there are over 450 ornamental fish species still on the official Australian importation list, with 200 individual species and 30 genera (encompassing 750 species) included on the live import list. Assessing the risks posed by all these species would require a much larger and more long-term study than the present review, so some restrictions on the scope of species to be covered was required. Of the 34 alien fish species that have established feral populations in Australian waters, 22 are thought to have come into the country via the ornamental fish trade (Lintermans 2004). Accordingly, this review only covers exotic aquarium or ornamental fish species established in Australian freshwater systems. The term established means that a breeding population exists somewhere in the wild in Australia (i.e., its future existence is not dependent on stocking). The terms ornamental and aquarium fish are both used to describe small fish kept as pets in home aquaria and so are taken to be synonymous. This review does not include marine species, or fish species introduced via ballast water discharge, or sports fish introduced into the wild in Australia including salmonids (which are covered as part of a parallel study). Neither does it include established exotic fish species such as roach, common carp, redfin perch, and tench, all of which are not aquarium, or ornamental species.

This report also covers only those species that are firstly confirmed as being established in the wild and, secondly, which were listed by Lintermans (2004). The reason for this is that some of the documented releases of ornamental fish into natural waterways in Australia may not result in successful establishment and there are likely to be other introduced species that have not become established at this point in time. One additional species (rosy barb, *Puntius conchonius*) not listed by Lintermans (2004) was added to this list because of advice from the Department of Agriculture, Fisheries and Forestry (DAFF) that it was now established in Australia. The 30 aquarium species reported as now being established in Australian waterways are listed in Table 1.1. The species covered by this report are indicated along with those present on the DEW live import list and thus currently allowed to be imported to Australia without a permit.

Table 1.1: List of ornamental freshwater fish species known to be established in the wild in Australia (source data [1]Kailola (2000), [2]Lintermans (2004)), including the species reviewed here, the *species currently on the Part 1 list of the DEW 'live import schedule' (i.e., not requiring a permit for importation) and the dates when the species were first recorded as being established in the wild (where known).

	Common name(s)	Scientific name	Sources	When first recorded as being established in the wild	Species reviewed in this report
Family Cichlidae					
1	Hybrid cichlid	*Labeotropheus/Pseudotropheus*	2	2001	Yes
2	Jewel cichlid	*Hemichromis bimaculatus*	1,2	2000	Yes
3	Victoria Burton's haplochromis	*Haplochromis burtoni*	1,2	1998	Yes
4	Black mangrove or Niger cichlid	*Tilapia mariae*	1,2	1978	Yes
5	Redbelly tilapia	*Tilapia zillii*	1,2	1980s	Yes
6	Blue tilapia	*Oreochromis aureus*	1		No
7	Mozambique tilapia or mouthbrooder	*Oreochromis mossambicus*	1,2	1970s	Yes
8	Oscar	*Astronotus ocellatus*	2	1998	Yes*
9	Three-spot cichlid	*Cichlasoma trimaculatum*	1,2	1998	Yes
10	Jack Dempsey	*Cichlasoma octofasciatum*	1,2	2004	Yes
11	Firemouth cichlid	*Thorichthys meeki*	1		No
12	Banded cichlid	*Heros severus*	1		No
13	Redhead cichlid	*Vieja synspila*	1		No
14	Red devil	*Amphilophus labiatus*	1,2	1992	Yes
15	Midas cichlid	*Amphilophus citrinellus*	1,2	1992	Yes
16	Convict cichlid	*Archocentrus nigrofasciatus*	1,2	1978	Yes
17	Blue acara	*Aequidens pulcher*	1,2	2000	Yes*
18	Green terror	*Aequidens rivulatus*	1		No
19	Pearl cichlid	*Geophagus brasiliensis*	1		No
Family Poeciliidae					
20	Green swordtail	*Xiphophorus hellerii*	1,2	1965	Yes*
21	Platy	*Xiphophorus maculatus*	1,2	1970s	Yes*
22	Sailfin molly	*Poecilia latipinna*	1,2	1969	Yes*
23	Guppy	*Poecilia reticulata*	1,2	1970s	Yes*
24	Caudo, one-spot livebearer	*Phalloceros caudimaculatus*	1,2	1970s	Yes
Family Osphronemidae					
25	Three-spot, blue or golden gourami	*Trichogaster trichopterus*	1,2	2000	Yes*

Family Cobitidae					
26	Oriental weatherloach	*Misgurnus anguillicaudatus/mizolepis*	1,2	1984	Yes
Family Cyprinidae					
27	Goldfish	*Carassius auratus*	1,2	1876	Yes*
28	Rosy barb	*Puntius conchonius*	1		Yes*
29	Sumatra barb	*Puntius tetrazona*	1		No
30	White cloud mountain minnow	*Tanichthys albonubes*	2	2003	Yes*

There is some confusion over the scientific names of some species owing to both changes in nomenclature and re-classification, as well as to uncertainty as to which of several closely related species are actually present in Australia. Wherever possible, the most recent scientific names recommended in FishBase[1] (Froese and Pauly 2006) are used here. Overall, 30 species of ornamental and/or aquarium fish have now been recorded in the wild (Table 1.1). This excludes gambusia (*Gambusia holbooki*), which while closely related to both the sailfin molly and guppy, is not considered an aquarium or ornamental species. In addition to the continuing taxonomic revision of species names, some species readily hybridise whereas breeding has created distinct strains for some species. This complicates impact assessment as the attributes of strains differ and those of hybrids can be expected to combine those of their constituent species in unpredictable ways.

The dates when the species were first recorded in Australia indicate that only 1 species was known to be present in the wild before 1950, but 2 were reported in the 1960s, 6 in the 1970s, 2 in the 1980s, 5 in the 1990s, and 6 in 2000s. This pattern of increase no doubt reflects the increased sampling effort and greater attention now applied to such fish, but it also suggests that releases and establishment are still occurring and it indicates that more species can be expected to be found over the next decade.

Several families of fish contain a large number of aquarium or ornamental fish species, none of which yet occur in Table 1.1. For example, there are many species of tetras belonging to the family Characidae that are popular aquarium fish. Similarly killifish belonging to the family Cyprinodontidae are also popular fish in freshwater aquaria. None of the species within either of these families (nor any of the exotic rainbow fishes in the families Melanotaenidae or Atherinidae) are listed as occurring in the wild in Australia even though these species can be expected to be widely present within freshwater aquaria. It seems unlikely that these fish have not been released into the wild (either inadvertently or deliberately) along with both cichlids and poeciliids.

1.3 Aims and objectives of the review

The overall aims of this study are to produce a report that firstly presents an objective understanding of the environmental, economic and social impacts (both positive and negative) of introduced aquarium fish species that have established wild populations

[1] Fishbase (Froese and Pauly 2006) is an international database that attempts to list all known fish species and provide a summary of information on all these species as far as is known. It also provides access to the literature on each species. The authors of the summaries for each species are not provided only a bibliography of the source material. It is therefore assumed that the species summaries reflect this material accurately and that they are updated as new information is published.

in Australia, and that secondly contributes to a cooperative and constructive approach to the management of introduced aquarium fish species, particularly for the protection of threatened native species and natural ecological communities.

The main tasks and objectives for this study are to:

- Map the current distribution of each established ornamental fish species in Australia.

- Assess evidence of ecological impacts associated with the established ornamental fish species and the methods used to assess these impacts.

- Based on this analysis, identify knowledge gaps and prioritise the need to fill these, and recommend alternative approaches to monitoring, impact assessment and research that will provide greater certainty with respect to actual impacts.

- Review the importation status of the 10 species currently on the Department of Environment and Water Resources's live import list (Part 1, Schedule: List of specimens taken to be suitable for live import – Environment Protection and Biodiversity Act 1999). This review will include a discussion of the implications of the 'do nothing' option, as against restricting some or all of these species from continued importation and sale in terms of likely ecological costs and benefits.

- Review control and eradication options for pest fish management in Australia. Given that few of the listed, established, ornamental fish have control and eradication strategies already devoted to them in Australia, DEW were keen to see what could be learned from overseas experience or experience associated with control of non-ornamental exotics such as salmonids and European carp. As part of this review, we also comment on the extent to which current or proposed control and eradication methods relating to exotic fish are socially acceptable.

- Estimate the value of the aquarium industry (including legal and illegal trade and subsidiary industries) and review studies of socio-economic cost/benefits carried out to assess other species in Australia. Assess the methods used to determine socio-economic costs associated with the impacts of exotic species and establish the type of economic modelling that should be applied to aquarium fish based on the quantity and nature of data available. Identify and prioritise the knowledge gaps that need filling in order to carry out such modelling.

1.4 Introduction to the species reviewed

Consideration of the phylogeny of the fish families represented by the species in Table 1.1 can provide a useful background to the interpretation of differences between the species. In particular, it allows consideration of some of the major adaptive differences characterising and distinguishing them, including feeding modes, parental care of eggs and water temperature requirements.

The species are classified into five families, three orders and two super-orders of fish (Fig. 1.2). The super-order Acanthopterygii differs from the Ostariophysi primarily in that its species have spiny as against soft fin rays. However, the Ostariophysi also possess a more specialised auditory sensory system based on adaptations associated with the air bladder (e.g., Weberian ossicles connecting the bladder to the inner ear). A number of species within the Ostariophysi are also capable of chemosensory communication based on their ability to detect chemicals such as pheromones and fright substances (see University of Liverpool Fish webpage at www.live.ac.uk/~rickl/Fisheries_Web).

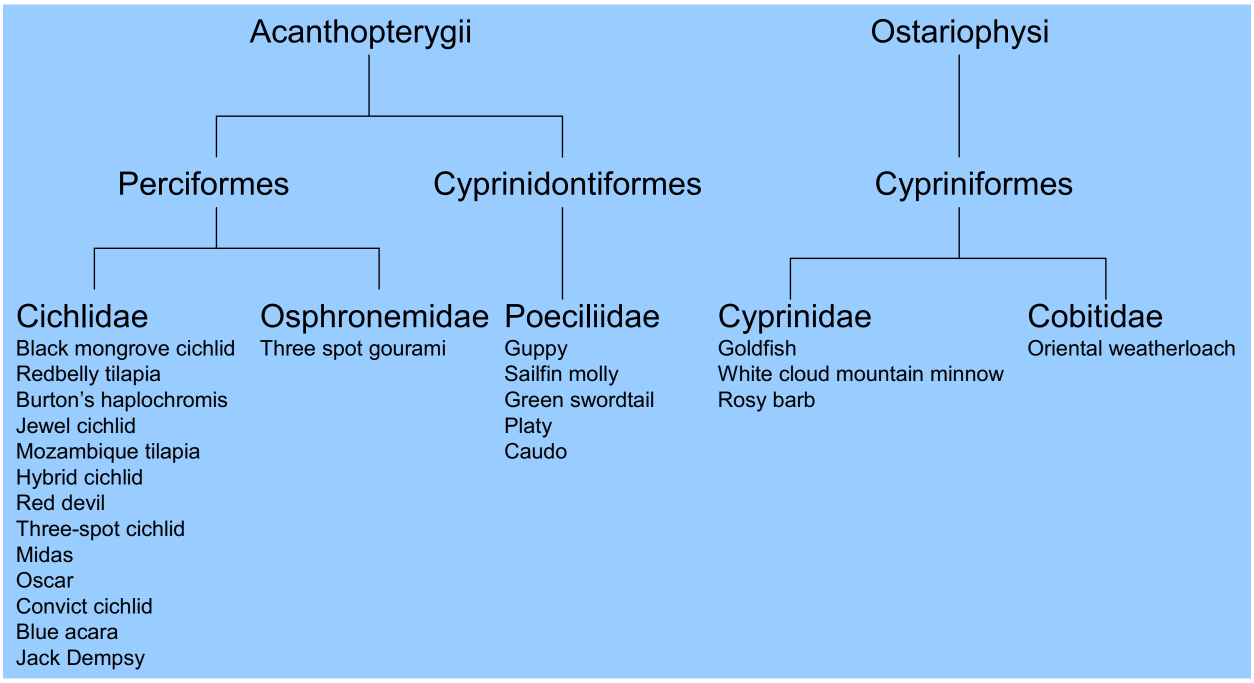


Figure 1.1: Phylogenetic relationships of the 23 species under review.

The spiny-rayed fish (Acanthopterygii) tend to dominate marine environments but some families (e.g., Cichlidae) have radiated widely within African lakes. In comparison, ostariophysian species, especially cyprinids, tend to occur more widely throughout freshwater habitats in Asia, Eurasia and in African rivers, but do not occur in South America. Possession of specialised auditory and chemical communication systems is thought to help explain the success of the ostariophysian fish in freshwater as against marine environments (University of Liverpool Fish webpage), but it is clear from the proliferation of cichlids in African lakes, and in many of the warmer freshwater environments in Asia, that many acanthopterygian species also have adaptations which suit them well to a wide range of freshwater environments.

Fish species belong to both the Cobitidae and Cyprinidae families lack true teeth, hence these ostariophysians are not specialised predators of highly mobile prey and instead possess pharyngeal teeth that allow them to 'masticate' food material. In general such feeding adaptations are associated with benthivorous behaviour and with feeding on plankton, plant matter and detritus. However, such fish are also able to feed on a wide range of benthic invertebrates and small fish. In contrast, all the acanthopterygian species listed in Table 1.1 possess teeth and can therefore be expected to actively prey on more mobile fauna throughout the water column.

The greatest number of species currently reported from the wild in Australia is from the family Cichlidae. Species within this family are generally characterised by one nostril positioned either side of the head and by nesting and/or parental guarding of the eggs and young. Many of the species within this family are also omnivorous, display aggressive behaviour and can remove vegetation from the substrate in the process of nest building (Midgalski & Fichter 1977). Approximately half of the cichlid species now present in the wild in Australia originated from south or north America, whereas the other half originated from Africa (Table 1.2). The taxonomy of this family is complex but Hougen (1994) provides a useful account of the main differences between groups that have evolved in the North versus South American continents and in Africa. In particular, he noted the greater number of taxa, the many reproductive strategies displayed and the wider size range for chichlids in Lake Tanganyika than in Lake Malawi and the very similar differences between cichlids from South versus North America. These differences imply that cichlids originating from Lake Tanganyika as against Lake Malawi and from South America as against North America can be expected to be much more specialised. In general, cichlids are warm-water fish found mostly in latitudes where summer water temperatures are above 20°C (Table 1.2).

Table 1.2: General characteristics of the 23 species of ornamental fish under review ([1]data from Fishbase).

Common name	Scientific name	Continent of origin[1]	Latitudinal temperature range[1] (°C)	Absolute range in temperature (degrees C)	Thermal classification based on latitudinal temperature range	Maximum fish size (cm)[1]	Spawning substrate or parental care of fry/eggs[1]
Family Cichlidae							
Hybrid cichlid	*Labeotropheus/Pseudotropheus*	Africa	------	--	------	----	-----
Jewel cichlid	*Hemichromis bimaculatus*	Africa	21-23	2	Stenotherm (warm)	14	-----
Victoria Burton's haplochromis	*Haplochromis burtoni*	Africa	20-25	5	Stenotherm (warm)	15	Mouth brooder
Black mangrove cichlid	*Tilapia mariae*	Africa	20-25	5	Stenotherm (warm)	40	Rocky substrates
Redbelly tilapia	*Tilapia zillii*	Africa	7-43	36	Eurytherm	40	Rocky substrates
Mozambique tilapia	*Oreochromis mossambicus*	Africa	8-42	34	Eurytherm	39	Mouth brooder
Oscar	*Astronotus ocellatus*	America	22-25	3	Stenotherm (warm)	40	Rocky substrate
Three-spot cichlid	*Cichlasoma trimaculatum*	America	21-30	9	Stenotherm (hot)	37	Nest guarder
Jack Dempsey	*Cichlasoma octofasciatum*	America	22-30	8	Stenotherm (hot)	25	Nest guarder
Red devil	*Amphilophus labiatus*	America	28-33	5	Stenotherm (hot)	24	Nest guarder
Midas cichlid	*Amphilophus citrinellus*	America	23-33	10	Stenotherm (hot)	24	Rock crevices
Convict cichlid	*Archocentrus nigrofasciatus*	America	20-36	16	Eurytherm	10	Range of substrate
Blue acara	*Aequidens pulcher*	America	18-23	5	Stenotherm (warm)	16	-----
Family Poeciliidae							
Green swordtail	*Xiphophorus hellerii*	America	22-28	6	Stenotherm (hot)	16	Livebearer
Platy	*Xiphophorus maculatus*	America	18-25	7	Stenotherm (warm)	6	Livebearer
Sailfin molly	*Poecilia latipinna*	America	20-28	8	Stenotherm (hot)	10	Livebearer
Guppy	*Poecilia reticulata*	America	18-28	10	Stenotherm (hot)	4	Livebearer
Caudo	*Phalloceros caudimaculatus*	America	20-24	4	Stenotherm (warm)	4	Livebearer
Family Osphronemidae							
Three-spot gourami	*Trichogaster trichopterus*	Asia	22-28	6	Stenotherm (hot)	15	Bubble nester
Family Cobitidae							
Oriental weatherloach	*Misgurnus anguillicaudatus*	Asia	10-25	15	Eurytherm	25	Range of substrates
Family Cyprinidae							
Goldfish	*Carassius auratus*	Asia	0-40	40	Eurytherm	60	Plant material
Rosy barb	*Puntius conchonius*	Asia	18-22	4	Stenotherm (warm)	14	Plant material
White cloud mountain minnow	*Tanichthys albonubes*	Asia	18-22	4	Stenotherm (warm)	4	-----

However, whereas most species have a somewhat restricted latitudinal native range (equating to an absolute temperature range of less than 10 degrees on the centigrade scale), the red belly tilapia and Mozambique tilapia both have a much wider latitudinal range than the other cichlids (i.e., equivalent temperature range of 34-36 degrees) and so can be expected to tolerate a much wider range of water temperatures. The three tilapia species (Genus *Tilapia* and *Oreochromis*) all grow to a large maximum size (40 cm), as does the three-spot cichlid and the oscar (Table 1.2). Other cichlids are somewhat smaller (10-25 cm maximum size).

The family Poeciliidae is characterised by species where fertilisation is internal, eggs are developed within the female body cavity, and the young are therefore born live. In general, the species now present in Australian waters are all from central and South America and are also warm-water fish likely to prefer water temperatures over about 20ºC (Table 1.2). They are mostly small fish with a maximum size of less than 16 cm (Table 1.2). This family includes the mosquito fish (genus *Gambusia*), which is an accepted pest in many places outside its native range, including in Australia.

The Osphronemidae differs from other families of fish in that its species possess a specialised breathing organ connected to their gill chamber and derived from adaptations of the swim bladder (Midgalski & Fichter 1977). This enables them to obtain oxygen by gulping air and allows them to colonise stagnant waters where other fish dependent on gills alone could not survive. The Osphronemidae also build nests made of small bubbles. The single exotic species now present in Australian waters (i.e., three-spot gourami) originated in southern Asia and, based on its native range, is also a warm-water species (Table 1.2). The three-spot gourami is also a relatively small fish.

The loaches (Cobitidae) also have a specialised organ for air breathing, but the labyrinthine organ used for this is connected to the intestine (Midgalski & Fichter 1977). Some species in this family (e.g., weather loaches) are known for their change in behaviour and increased activity when the barometer drops. This is thought to be related to the effects of changing air pressure on their labyrinthine organ. Although the oriental weatherloach originated in Asia, its latitudinal range is much greater than that of the three-spot gourami, and it can be expected to occupy a wider latitudinal range in Australia and to tolerate lower water temperatures than either the Poeciliidae or Osphronemidae.

The Cyprinidae are a diverse family and as with the Cobitidae lack teeth and spiny fin rays. They generally possess barbels which they use to detect prey or food within or on the bed of the waterbody they inhabit. This family contains carp (*Cyprinus carpio*), roach (*Rutilus rutilus*) and tench (*Tinca tinca*), which also occur in the wild in Australia. All three species of ornamental cyprinids now in Australia (Table 1.2) are from Asia and except for the goldfish have a similar temperature range to many of the

Poeciliids. Hence, they too can be expected to prefer relatively warm-waters (i.e., over 20ºC).

In general, the species of ornamental fish in Table 1.2 can be classified by the water temperature range associated with their native latitudinal range into either stenotherms (i.e., they occur where the absolute temperature range is small to moderate, e.g., 2-10 degrees on the centigrade scale) or eurytherms (i.e., they occur where the absolute temperature range is wide e.g., 15-40 degrees on the centigrade scale). Eurytherms can by definition cope with a wide range of water temperatures and therefore have a potentially wider geographic distribution within Australia than stenotherms. Stenotherms are associated with a more restricted temperature range, which may be relatively hot or cold, or somewhere between these two extremes. Of the stenotherms listed in Table 1.2, those species associated with relatively hot maximum water temperatures (i.e., 28-33ºC) might be expected to have a more northerly potential distribution in Australia than the species associated with comparatively warm maximum water temperatures (i.e., 22-25ºC).

These differences among the families, and species within them, provide the main points of difference between aquarium fish and other exotic fish in Australia such as salmonids, perch and carp. In general, the aquarium fish species are smaller and require warmer waters than the sports fish and none of them are specialised piscivores. However, the poeciliids, which are the smallest species, have some traits in common with mosquitofish (*Gambusia affinis* and *G. holbrooki*) both of which are known to have affected native fish in other parts of the world, including Australia (Arthington & Lloyd 1989; Moore et al. 2002; Morgan et al. 2004). Although the aquarium fish species now established in the wild in Australia display a number of differences to exotic sports fish and fish introduced for mosquito control, international experience with exotic fish introductions indicates that a careful, species-by-species analysis of evidence for impacts in the wild is required. Such information is a pre-requisite for the future management of these species.

2. Distribution of established aquarium fish species in Australia

2.1 Introduction

Before examining the known distribution of the 23 ornamental fish species established in Australia to date, it is important to define the Australian aquatic environment being considered. For the purposes of this study, the species of ornamental fish being investigated are primarily freshwater fish and do not include any saltwater ornamental species utilised in seawater aquaria. The environment in which the target species could occur therefore includes all the freshwater habitats within the continental landmass of Australia and the adjacent island of Tasmania. We have not included freshwater habitats in Australian offshore territories such as the Cocos Islands, Torres Straits Islands, Lord Howe Island, Norfolk Island or Christmas Island. Although inland saline and brackish water lakes are included, coastal saltwater habitats in harbours and around the coastline are excluded. This distinction is practical rather than ecological as some freshwater species may well become adapted to brackish and saltwater habitats. For example, *Gambusia affinis* is an example of a small freshwater fish that readily adapts to full strength saltwater and which is now abundant among mangrove swamps in a number of New Zealand harbours (Mitchell 1985). Conversely, some saltwater species such as the dart goby (*Parioglossus marginalis*) are capable of inhabiting freshwater habitats (McDowall 2001).

There are several features of Australian freshwaters that, in our view, make Australia much more vulnerable to invasion by ornamental fish than neighbouring countries such as New Zealand to the southeast or Papua New Guinea to the north. Firstly, the Australian continent covers a vast latitudinal range and encompasses a wide range of climate zones (e.g., tropical, subtropical, temperate and arid). Secondly, there is a wide diversity of habitat types for freshwater fish in Australia, including rivers, lakes, streams, estuaries, billabongs, wetlands, brackish lakes, floodplains and thermal springs. This combination of broad climatic range and high habitat diversity means that there is a much greater chance of ornamental fish becoming established somewhere in Australia than in a more temperate country such as New Zealand or in a more tropical and latitudinally compressed country such as Papua New Guinea. Moreover, the low gradient of much of inland Australia means that there are large numbers of ponds and small lakes. There is therefore a greater likelihood that the environmental requirements of at least some exotic fish species will be met somewhere in Australia. Hence the risk of ornamental fish species becoming established somewhere in Australian waters and subsequently spreading within and among them is much greater than in neighbouring countries. Over time, the species that establish founder populations in a restricted habitat may become adapted to a wider range of conditions and acquire the ability to spread well beyond what would currently be considered their original habitable range (e.g., Arthington 1991).

The detection of new incursions of exotic species is therefore more important for Australia than for neighbouring countries, and distribution mapping will be required to determine the location and rate of spread of all species. In this chapter we present the known data on the distribution of the 23 ornamental fish species currently established in Australia. Maps provide a basis for identifying where incursions have already occurred and a baseline for future monitoring of species' spread. It should be noted that the distributions portray the region(s) within which reproducing populations have been discovered and not locations where their absence can be confirmed. Furthermore, it is acknowledged that many inland waters have not been sampled adequately to date, and that the status of the 23 ornamental fish species cannot be presented for these waters. These omissions reflect the paucity of data on fish occurrence in Australian waters and emphasize the need for a coordinated national database and sampling programme to record and hence monitor exotic fish distributions. These distribution maps therefore need to be interpreted cautiously and although some trends are apparent, the reasons for them are speculative and will require further evaluation.

2.2 Collection of data

The main method for obtaining distribution data for the 23 established ornamental fish in Australia was through reviewing reports and scientific journal articles or syntheses of these (e.g., by the Australian Society of Fish Biologists or ASFB). Some of the standard natural history texts that cover the fauna were also consulted (Merrick and Schmida 1984; Allen et al. 2002). In addition, we elicited data from individuals in Federal, State and Local Government agencies, regional bodies and universities whom we considered most likely to have access to such data, or who could direct us to more appropriate sources. Many of these individuals were identified through their publications on exotic fish, although networks of contacts were also utilised to obtain additional contacts. Those we contacted are listed in Appendix 13.1

When sourcing distribution data from individuals we sent out two surveys via email. The preliminary survey was a simple questionnaire aimed at obtaining a basic understanding of which states each species was present in and of gauging who, among those contacted, were willing to provide us with more detailed distributional data. A copy of this survey is shown in Appendix 13.2. A second questionnaire was then sent out to those who volunteered to provide us with more detailed distribution data, and was aimed at obtaining a more precise indication of which water bodies each species occurred in within each state and to gather data on the type of habitat in which they were found. A copy of this questionnaire is shown in Appendix 13.3. The initial questionnaire was distributed to 32 people across Australia known to have some knowledge of aquarium fish in the wild. A 72% response rate was achieved but many respondents felt that the questions were too specific for them to be able to answer adequately and rapidly.

Once we had collated the detailed distribution data we incorporated this information into maps as used by the ASFB and displayed on their website. We chose this format as it was likely to be familiar to many of the readers of this report. It was also one that did not reveal specific locations. This was considered important by the project team because publication of such information might result in problems for some land owners and raise the possibility that some individuals might exploit this information (e.g. to collect exotic fish from the locations either for profit or to transfer them to new locations). Furthermore, the location of populations was often limited to either general areas (see Table 2.1) or to streams and rivers rather than to identifiable reaches within catchments. Hence it was not possible to provide more accurate indications of the location of species such as grid references on maps or GIS positions.

The detailed information on known locations of populations of exotic ornamental fish is provided in Table 2.1 Four additional ornamental fish species not included in the review list were included in this mapping survey. These are the pearl cichlid, firemouth cichlid, green terror and banded cichlid (Table 2.1). The distributions of all these species based on the data collated are shown in Figures 2.1-2.27. The only ornamental species known to be present in the wild (Table 1.1) but for which no distributional data were found are the blue tilapia (*Oreochromis aureus*), the redhead chichlid (*Vieja synspila*) and the Sumatra barb (*Puntius tetrazona*).

Table 2.1: Summary of the locations in which the ornamental fish established in Australian waters are currently known from.

Scientific name	**Common name**	**Locations found in Australia**	**Information source**
1. Hybrid cichlid	*Labeotropheus/Pseudotropheus*	Hazelwood power station (Vic)	ASFB (2001)
2. Jewel cichlid	*Hemichromis bimaculatus*	Rapid Creek in Darwin (NT); Ross River (northern Qld)	ASFB (2003b); A. Webb (pers. comm.); D. Wilson (pers. comm.)
3. Victoria Burton's haplochromis	*Haplochromis burtoni*	Ross River in northern Qld & Hinze Dam (south-east Qld)	Arthington et al. (1999); ASFB (2003b); A. Webb (pers. comm.)
4. Black mangrove cichlid	*Tilapia mariae*	Cairns area, Barron, Ross, Johnstone, Burdekin, Mulgrave and Russel Rivers (Qld); Hazelwood power station, Eel Hole Creek., Latrobe River (Vic.); Lake Burley Griffin Canberra (ACT)	Cadwallader et al. (1980); Arthington et al. (1999); McKenzie et al. (2000); ASFB (2001); Allen et al. (2002); ASFB (2003b), ASFB (2004b); A. Webb (pers. comm.)
5. Redbelly tilapia	*Tilapia zillii*	Chapman River near Geraldton (WA)	Arthington et al. (1999)
6. Blue tilapia	*Oreochromis aureus*	No data obtained	
7. Mozambique tilapia	*Oreochromis mossambicus*	Brisbane dams, Boyne River including Boondooma Dam, tidal Creeks around Townsville, Cairns, Atherton Tableland, Endeavour R. & Port Douglas; Barron, Ross, Mulgrave & North & South Johnstone and Pine Rivers, (Qld); Gascoyne, Lyons, Milnilya & Chapman Rivers in the Pilbara Drainage & limestone caves Exmouth (WA)	Arthington et al. (1984); Arthington & Milton (1986); Bludhorn & Arthington (1990b); DPIQ (2000); Allen et al. (2002); Low (2002); AFSB (2003b); ASFB (2003c); Lintermans (2004); Morgan et al. (2004); A. Webb (pers. comm.);
8. Oscar	*Astronotus ocellatus*	Ross River & creeks around Cairns (northern Qld)	Arthington et al. (1999); ASFB (2003b); A. Webb (pers. comm.)
9. Three-spot cichlid	*Cichlasoma trimaculatum*	Hinze Dam (south-east Qld)	Arthington et al. (1999)
10. Jack Dempsey	*Cichlasoma octofasciatum*	Angourie (northern NSW)	M. Lintermans (pers. comm.)
11. Firemouth cichlid	*Thorichthys meeki*	Ross River (northern Qld)	Arthington et al. (1999); ASFB (2003b); A. Webb (pers. comm.)
12. Banded cichlid	*Heros severus*	Ross River in (northern Qld)	Arthington et al. (1999); ASFB (2003b); A. Webb (pers. comm.)

13. Redhead cichlid	*Vieja synspila*	No data obtained	
14. Red devil	*Amphilophus labiatus*	Ross River (northern Qld); & Hinze Dam (south-east Qld); Hazelwood pondage, LaTrobe Valley (Vic)	Arthington et al. (1999); ASFB (2001); A. Webb (pers. comm.)
15. Midas cichlid	*Amphilophus citrinellus*	Ross River (northern Qld)	ASFB (2003b); A. Webb (pers. comm.); M. Lintermans (pers. comm.)
16. Convict cichlid	*Archocentrus nigrofasciatus*	Ross River & streams around Townsville (northern Qld); Hazelwood power station, Eel Hole Creek, LaTrobe River (Vic.)	Cadwallader et al. (1980); Arthington et al. (1999); ASFB (2001); Allen et al. (2002); ASFB (2003b)
17. Blue acara	*Aequidens pulcher*	Creeks in Brisbane & Leslie Dam (south-east Qld); Hazelwood power station (Vic)	Arthington et al. (1999); ASFB (2001)
18. Green terror	*Aequidens rivulatus*	Ross River (northern Qld)	Arthington et al. (1999); ASFB (2003b); A. Webb (pers. comm.)
19. Pearl cichlid	*Geophagus brasiliensis*	Quarry & ornamental pool at Rockhampton & Bajool (Qld)	Arthington et al. (1999)
20. Green swordtail	*Xiphophorus hellerii*	Streams and rivers around Brisbane, Gladstone, between Maryborough & Cairns, Barron & Ross Rivers (northern Qld); Lake Ainsworth near Lennox Head & Burringbar Creek northern NSW (NSW); town Billabong in Nhulunbuy, dam at Alice Springs & Gunn Point and waters in the vicinity of Darwin (NT); Irwin River (WA).	Arthington et al. (1983); Morgan & Gill (2001); Allen et al. (2002); ASFB (2003a); ASFB (2003b); Morgan et al. (2004); A. Webb (pers. comm.); A. Moore (unpublished data); D. Wilson (pers.comm.), Northern Land Council (www.nlc.org.au).
21. Platy	*Xiphophorus maculatus*	Streams, swamps & drains around Brisbane, Calliope, Burrum Ross, Barron, Russell, Mulgrave, Tully, Johnstone & Babinda Rivers & Behana, Peewee, Louisa & Harley Creeks (northern Qld); town billabong in Nhulunbuy & Rapid Creek Darwin (NT).	Arthington et al. (1983); Arthington et al. (1999); Allen et al. (2002); ASFB (2003b); D. Wilson (pers.comm.); A. Webb (pers. comm.).
22. Sailfin molly	*Poecilia latipinna*	Streams and rivers around Brisbane & Harvey Bay, Ross River (northern Qld), waters in the vicinity of Darwin (NT).	Arthington et al. (1983); Arthington et al. (1999); Allen et al. (2002); ASFB (2003b); M. Lintermans (pers. comm.)

			Northern Land Council (www.nlc.org.au).
23. Guppy	*Poecilia reticulata*	Coastal drainages of Qld from Cairns to Brisbane, including the Burnett, Black Alice, Ross, Herbert, Fitzroy, Barron, Murray, Mossman, Mulgrave, Moresby & North & South Johnstone Rivers, Alligator & Crystal Creeks, Gustav Creek Magnetic Island, ponds & streams in Charters Towers (Qld); Billabong in Nhulunbuy, Railway Dam, Leanyer Swamp & Sadgroves Creek Darwin (NT); Roadside pool in Pilbara Drainage (WA); Coastal drainages of northern NSW	Arthington et al. (1983); Arthington et al. (1999); Allen et al. (2002); ASFB (2003a); Morgan et al. (2004); A. Webb (pers. comm.); D. Wilson (pers.comm.).
24. Caudo	*Phalloceros caudimaculatus*	Swamps & drains around Perth, Swan-Avon Rivers; Canning River (WA); Northern NSW & Sydney	Arthington et al. (1999); Allen et al. (2002); ASFB (2003a); Morgan et al. (2004); Rowley et al. (2005)
25. Three-spot gourami	*Trichogaster trichopterus*	Ross River & lower floodplain of the Burdekin River, Sheepstation Creek (northern Qld)	Arthington et al. (1999); ASFB (2003b); A. Webb (pers. comm.)
26. Oriental weatherloach	*Misgurnus anguillicaudatus*	Hazelwood power station, LaTrobe catchment & Yarra, Maribyrnong, Patterson, Campaspe, Don, Ovens & Murray Rivers, Corhanwarrabul, Nine Mile, Broken, Koonung, Ruffey & Dandenong Creeks (Vic); Mountain Creek Murrumbidgee catchment, Murray, Wingecarribee, Queanbeyan, Peak, Wollondilly, Cox's Edwards, Neimur, Hawkesbury-Nepean Rivers & Lake Eucumbene, Tuppal Creek (NSW); Common in lowland streams including the lower Cotter, Paddy, Molonglo, Gudgenby, Queanbeyan and Ginninderra Rivers, Goorman, Halls, Tuggeranong Creeks, Lake Burley Griffin (ACT)	Allen (1984); Arthington et al. (1999); ASFB (2001); Koster et al. (2002); ACT (2002); ASFB (2003a); M. Lintermans (pers. comm.).

27. Goldfish	*Carassius auratus*	Fitzroy, Dawson & Burnett Rivers in northern Qld to NSW including most coastal & inland waters of NSW, Vic. & southern Qld; Coastal drainages of south western WA between Moore, Vasse & Blackwood Rivers, Canegrass Swamp & Bromus Dam (WA); common in lowland streams (ACT); Western Plateau of SA & Coopers Creek Lake Eyre drainage (SA)	Arthington et al. (1999); Allen et al. (2002); Morgan et al. (2004); M. Lintermans (pers. comm.)
28. Rosy barb	*Puntius conchonius*	Streams in and south of Brisbane (Qld); Margaret River area Western Australia	Allen et al. (2002); Arthington et al. (1999); ASFB (2006)
29. Sumatra barb	*Puntius tetrazona*	No data obtained	
30. White cloud mountain minnow	*Tanichthys albonubes*	Creek in Brisbane (Qld); Green Point Creek Central Coast (NSW)	ASFB (2003b); ASFB (2003c)

2.3 Maps of species distributions

Figure 2.1: Distribution of hybrid cichlid (*Labeotropheus/Pseudotropheus* cross).

Figure 2.2: Distribution of jewel cichlid (*Hemichromis bimaculatus*).

Figure 2.3: Distribution of Victoria Burton's haplochromis (*Haplochromis burtoni*).

Figure 2.4: Distribution of black mangrove cichlid (*Tilapia mariae*).

Figure 2.5: Distribution of redbelly tilapia (*Tilapia zillii*).

Figure 2.6: Distribution of mozambique tilapia (*Oreochromis mossambicus*).

Figure 2.7: Distribution of oscar (*Astronotus ocellatus*).

Figure 2.8: Distribution of three-spot cichlid (*Cichlasoma trimaculatum*).

Figure 2.9: Distribution of Jack Dempsey (*Cichlasoma octofasciatum*).

Figure 2.10: Distribution of firemouth cichlid *(Thorichthys meeki).*

Figure 2.11: Distribution of banded cichlid (*Heros severus*).

Figure 2.12: Distribution of red devil (*Amphilophus labiatus*).

Figure 2.13: Distribution of midas cichlid (*Amphilophus citrinellus*).

Figure 2.14: Distribution of convict cichlid (*Archocentrus nigrofasciatus*).

Figure 2.15: Distribution of blue acara (*Aequidens pulcher*).

Figure 2.16: Distribution of green terror (*Aequidens rivulatus*).

Figure 2.17: Distribution of pearl cichlid (*Geophagus brasiliensis*).

Figure 2.18: Distribution of green swordtail (*Xiphophorus hellerii*).

Figure 2.19: Distribution of platy (*Xiphophorus maculatus*).

Figure 2.20: Distribution of sailfin molly (*Poecilia latipinna*).

Figure 2.21: Distribution of guppy (*Poecilia reticulata*).

Figure 2.22: Distribution of caudo (*Phalloceros caudimaculatus*).

Figure 2.23: Distribution of three-spot gourami (*Trichogaster trichopterus*).

Figure 2.24: Distribution of oriental weatherloach (*Misgurnus anguillicaudatus*).

Figure 2.25: Distribution of goldfish (*Carassius auratus*).

Figure 2.26: Distribution of rosy barb (*Puntius conchonius*).

Figure 2.27: Distribution of white cloud mountain minnow (*Tanichthys albonubes*).

2.4 Distribution patterns and their implications

Queensland contains the highest number of established ornamental species including six species that have not been reported from there before (i.e., midas cichlid, oscar, three-spot cichlid, Victoria Burton's haplochromis, three-spot gourami and rosy barb. Only 5 out of the 23 listed species have not been reported from Queensland so far (i.e., redbelly tilapia, Jack Dempsey, *Labeotropheus/Pseudotropheus* cross cichlid, oriental weatherloach, one-spot live bearer). In contrast, the only listed ornamental species reported from Tasmania is the goldfish.

These latitudinal differences in ornamental fish distribution in the wild are not unexpected given that many of the listed species are tropical, warm-water fish. However, the preponderance of ornamental species in Queensland is not reflected in the Northern Territory or in the northern region of Western Australia. This may reflect the low potential for introduction of exotic fish in more remote (northwestern) areas, but it may also reflect the paucity of fish surveys in these regions

Goldfish have been reported from the largest area in Australia (Fig. 2.25). Not only is this species present in 6 out of the 8 States and Territories but it occurs in many waters throughout the entire south-eastern region of Australia. No records were reported for the northern half of the continent.

Mozambique tilapia (Fig. 2.6), guppy (Fig. 2.21), and platy (Fig. 2.19) are present over large geographic areas but these areas are confined to coastal regions.

Of the species, apart from goldfish that are typically found in temperate regions, oriental weatherloach was the most widespread in Australia in terms of its reported distribution (Fig. 2.24). Populations are now present in large areas of southern NSW, ACT and Victoria. One population of the white cloud mountain minnow was reported from Brisbane and another occurs further south (Fig. 2.27).

There are a number of species that, despite occurring within discrete locations, have been reported from a wide latitudinal range. These include red devil (Fig. 2.12), convict cichlid (Fig. 2.14), black mangrove cichlid (Fig. 2.4), Victoria Burton's haplochromis (Fig. 2.3), blue acara (Fig. 2.15), sailfin molly (Fig. 2.20) and the green swordtail (Fig. 2.18). These species all have established populations in both the tropics and in temperate regions of Australia. These findings belie the temperature ranges that many of these species occur within in their native range, indicating that matching environmental tolerances with climate (i.e., bioclimatic matching) is not always a reliable way of predicting the potential distribution of these species.

Species with a restricted latitudinal range but with a wide longitudinal range include the one spot live bearer (Fig. 2.22) and jewel cichlid (Fig. 2.2).

The remaining species have only been reported from a few restricted locations at present. They include midas cichlid (Fig. 2.13), oscar (Fig. 2.7), red belly tilapia (Fig. 2.5), three spot cichlid (Fig. 2.8), Jack Demsey cichlid (Fig. 2.9), hybrid cichlid (Fig. 2.1), three spot gourami (Fig. 2.23), rosy barb (Fig. 2.26), firemouth cichlid (Fig. 2.10), pearl cichlid (Fig. 2.17), green terror (Fig. 2.16) and banded cichlid (Fig. 2.11).

It is clear that there is strong human involvement in the way ornamental fish are dispersed in Australia (Lintermans 2004) and the distributional data presented here confirm this. Liberations of exotic fish are often concentrated around metropolitan areas because these are areas where the fish are kept in aquaria and local waters are readily accessed. In contrast, the spread of such species into more remote areas is typically limited (Arthington et al. 1999; Kailola 2000; Bomford and Glover 2004), mainly because of the cost and inconvenience of transporting them there. Human vectors can explain much of the clustering of these species close to major urban centres in Australia. For example, 16 species of ornamental fish are known to be established near Townsville, 10 near Brisbane, 5 near Cairns, 4 near Darwin, 4 in northern NSW, and 3 near Sydney (Figure 2.28).

No clusters of the ornamental species occurred away from major urban centres. More sampling is typically carried out close to such centres than far away from them and the actual number of species found may reflect this. The proximity of the clusters in Figure 2.28 to major urban centres indicates that human dispersal is a major factor in the spread and establishment of these species and that public education is needed to counter this.

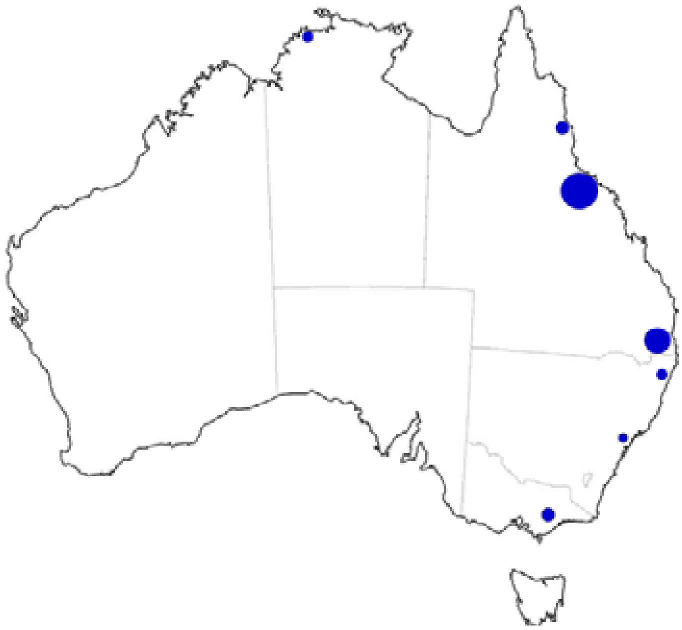


Figure 2.28: Locations where large numbers of ornamental fish species have become established in Australia. (The size of the circles reflects the number of species known to be established and ranges from 3 in Sydney to 16 in Townsville).

More incursions of ornamental fish would be expected near the warmer, northern cities of Darwin, Cairns, Townsville and Brisbane, than near southern cities and exotic fish can be expected to be more successful at surviving and breeding in the warmer waters of the north. This is borne out by the data in Figure 2.28. However, the presence of 16 species near Townsville compared with only 5 in Cairns and no clusters in Mackay and Rockhampton is surprising. This is likely to reflect a bias in sampling coverage (i.e., proximity to James Cook University), but could also indicate that aquarium species have had a longer history in Townsville, or are now more popular there than in other northern centres.

The absence of species clusters near Perth, Adelaide, Canberra, Melbourne and Hobart may reflect their more southern location and the failure of ornamental species to establish in colder waters, but it may also reflect a lack of sampling and/or a reduced interest in aquarium species in these centres.

Although a warmer climate and human vectors can account for much of the overall distribution pattern for the clusters of ornamental fish species in Australia, the presence of a cluster of 5 species in western Victoria was associated with the thermal discharge from a power station (Arthington et al. 1999). This tends to support the view that although many ornamental species may have been released into natural waters close to major urban centres, their establishment has probably been constrained by a lack of suitable habitat (i.e., warm waters) in many southern regions.

2.5 Future distribution mapping

Overall, the distributional data on the established populations of ornamental fish suggests that the risk of these species becoming established is greater in the north where water temperatures are warmer than in the south. However, some warm-water species have become established in temperate as well as tropical regions. If the main environmental tolerances for the 23 ornamental species were known it would be theoretically possible to match these with environmental conditions and to develop maps of the potential spread of each species throughout Australia. Koehn (2004) carried out such a prediction for European common carp using CSIRO's CLIMEX model. This model is based on air temperature data collected from monitoring stations throughout Australia and data on the thermal tolerance range for target alien species[2]. While the projected distribution map for the distribution of carp was patchy due to the lack of coverage of air monitoring stations in Australia, the outputs of CLIMEX were useful in that they suggested that this species was capable of surviving in most parts of Australia. However, Koehn (2004) noted that a better solution would be to use a similar model based on water temperatures rather than air temperatures. In reality, this approach is probably impractical for many of the ornamental species found in Australia because their key environmental tolerances are not well known and corresponding data on the aquatic environment are also sparse. Furthermore, the temperature range occupied by a species in its natural range may not indicate its physiological tolerance and may reflect other factors such as the effects on its distribution of predation and competition by other species or physical habitat constraints. Maps of the predicted geographical distributions of the species studied here have been prepared using the known data on temperature tolerances (Bomford & Glover 2004). These provide a guide to potential distribution, but those species that can hybridise with other species may produce viable progeny that have wider tolerances (c.f., Arthington 1991, Mather & Arthington 1991) and, over time, strains of introduced species may become more tolerant of a wider range of environmental conditions.

A further complication to predicting fish distributions from environmental tolerances is provided by the occurrence of barriers to fish movement. These may restrict their spread despite the presence of suitable habitat up or downstream. Barriers not only include physical impediments to upstream movement such as culverts, falls and dams, but reaches with higher water velocities that may be impassable to species that are small, or which have a poor swimming ability. Similarly, higher salinity reaches connecting freshwater streams may prevent the spread of species with low salinity tolerances. For these reasons, environmental-tolerance based predictions of the potential spread of ornamental fish species is likely to be of limited value and other methods need to be developed to map fish distributions and to determine the potential spread of exotic fish species.

[2] CLIMEX has been used for predicting Cane Toad distributions in Australia (Koehn, 2004).

Of these methods, the creation of a fish database is likely to prove the most useful in the long term. The United States Geological Survey (USGS) maintains an interactive database portraying the distribution of non-native, exotic fish in the United States (http://nas.er.usgs.gov/queries/deafult.asp). It can be interrogated to provide information on exotic fish reports and distributions for single or multiple species at scales ranging from watersheds upwards. It also provides an 'Alert' system which flags the location of recent new incursions of exotic species and so provides a nationwide overall coordination of issues related to exotic species. Such a centralised database is an essential tool for managing exotic species.

In New Zealand, the National Institute for Water and Atmospheric Research (NIWA) owns and manages a freshwater fish database. This is outlined in Richardson (2005) and has proved highly effective for mapping fish distributions and for generating predictions and hypotheses about the habitats and geographic spread of various fish species. The database was set up over two decades ago and can be added to or interrogated easily by external organisations such as other government agencies or consultants that have particular questions they may wish to have answered with respect to the distribution of fish in New Zealand's waterways. It is now linked to GIS layers providing data on geology, topography and flow regime. A similar system owned and maintained by a central federal agency in Australia such as DEW or DAFF would be an extremely useful tool for collating records, mapping fish distributions and therefore for monitoring the spread of established exotic fish species in Australian waterways. This would facilitate the prioritisation of research, monitoring, control and eradication of such species and it would provide national coordination to ensure that control programmes in one state are not compromised by a lack of action in another region or State.

Queensland is one of the few, if not the only state (to our knowledge), to use an exotic fish database and mapping programme as part of their exotic fish management strategy. The use of PestInfo as a database to accommodate pest fish distribution data was suggested by Mackenzie (2003) as part of Queensland strategy for controlling and managing pest fish species in the state.

The distribution data on pest fish, including established ornamental species, is incorporated into the PestInfo database, which is a GIS database (pers. comm.. Mr T Chen, Queensland Department of Primary Industries and Fisheries or QDPI). Data are supplied to QDPI&F largely by regional management groups (essentially catchment management organisations). Data include the types of pest fish species, the GPS position where they are found, habitat information and the name of the property owner (if found on private property). No biological or taxonomic specialist advice is incorporated into quality assurance of these data before entry into the database, so some identifications may be suspect and the robustness of the data may be questionable in certain cases. Regional management organisations are, however, given good keys for identifying pest fish species and should be able to readily identify pest species rated as high priority. One limitation of this arrangement is that species that are more difficult to identify or those that have only recently been introduced might be confused with other species or overlooked altogether. A further limitation on the use or access to this information is that regional management groups supplying the data have a copyright on the information, mainly to protect local councils and private property owners from problems. This copyright regulation prevents information being disclosed to third parties, which may limit its usefulness as a tool for research organisations to develop impact hypotheses and research the ecology of established ornamental species.

QDPI&F also have their own system in place for assessing the distribution of pest fish species in Queensland. This is the Annual Pest Fish Distribution Survey. For this, inland regions of the state are divided up into 50 x 50 km grids, while the coastal regions are divided up into 18 x 18 km grids. Before monitoring takes place, each grid square is evaluated in terms of the type of pest species already present, their priority rating (class 1 through to 3) and their density. This information is drawn together by members of each regional management group as well as fisheries scientists. A fish taxonomy / ecology specialist is on hand to assess the information gained for its validity. Through this procedure, QDPI&F can develop targeted research and monitoring, as well as control and eradication programs. Distribution data collected through this processes is held by QDPI&F, though it is not clear whether it is also incorporated into the PestInfo database. The data are copyrighted by QDPI&F, so are available for use by outside organisations or individuals at their discretion. Information from this database was not used in our distribution mapping because of budgetary and IP constraints, but can be expected to provide a finer-scale indication of the distribution of ornamental fish species in Queensland.

There are several experts specialising in freshwater fish biology and invasive fish biology in Australia who also hold their own databases. From our discussions as part of this study, we are aware of databases held by Alan Webb (James Cook University) for a variety of exotic fish occurring in far North Queensland and databases held by Brad Pusey and Angela Arthington (Griffith University) for South-East Queensland that incorporate distribution data for both native and exotic fish species. We have also been told by Mick Howland from HydroTasmania that a fish distribution database exists for Tasmania, which is purportedly owned and operated by Peter Davies at the University of Tasmania. There are probably other researchers who hold similar fish distribution databases in other parts of Australia, though only Alan Webb advised us that he was keen to set up a robust database that was more accessible than that held by the state government (QDPI&F). Alan was concerned that lack of access to such information could actually hamper research and control and eradication efforts with respect to exotic fish in Queensland. It is clear that if a centralised exotic fish distribution database was to be set up for Australia, State agencies would need to buy into the process and see it as a useful tool as well a key component of their management and decision-making procedures. They are likely to be both the major contributors as well as the main beneficiaries of such a system.

2.6 Recommendations

1. Confirm identifications of species over which there is confusion (e.g., hybrid cross cichlid, oriental weatherloach) and ensure that good keys are available to aid the identification of aquarium fish species in the field.

2. Develop better diagnostic methods for identifying exotic fish being imported into Australia.

3. Confirm the presence in the wild for the species reported to be present but for which no geographic data could be obtained.

4. Confirm the presence of breeding populations for species with very limited distributions and assess their risk of spread.

5. Investigate the feasibility of establishing a national data base for recording the distribution and spread of freshwater ornamental fish in Australia.

6. Recommend further investigation of the causes of 'hot spots' for species incursions in northern Queensland and where important help to develop targeted public relations campaigns to counter species' introductions. This study would need to address the relationship between propagule pressure and introduction pathways along with other potential causes of such hot-spots.

7. Identify isolated incursions of species considered to pose threats and rapidly determine the feasibility of their eradication to prevent further spread.

3. Review of impact assessment methodologies

3.1 Introduction

There are diverse perceptions in Australia about the extent and nature of impacts of exotic fish species on the environment. The establishment of some exotic fish (e.g., trout) is seen by many as beneficial whereas the establishment of other species (e.g., common carp) is regarded as detrimental. Such generalisations are value judgments rather than scientific assessments (Rosenweig 2001; Slobodkin 2001 *cited in* Lodge and Shrader-Frechette 2003) and a number of fish ecologists have recently expressed concerns over the potential for some, but not all, exotic fish to cause ecological problems in Australia. These ecologists provide a more objective view of impacts and advocate the use of a scientific approach rather than value judgements to test their concerns and to provide objective information on the potential for impacts to occur in the wild.

The need for valid information on impacts is especially important for aquarium or ornamental fish species in Australia because a number of these species are now established in the wild (see chapter 2) and blanket control would be very costly and may not be required. Furthermore, differences in perceived impact arising from a lack of information can result in conflict between opposing stakeholder groups and increase pressure on government and resource management agencies to develop effective policy and actions. However, it is always difficult to deliver effective policy and management where the facts are unclear and any management attempts to prioritise control and eradication actions for certain species will be undermined if there is little objective information on impacts, or if the potential benefits of proposed control measures are unknown (Bomford and Tilzey 1996).

Uncertainties about the impacts of ornamental fish species could also lead to resistance by stakeholders to contribute to the management of the issues. Unless there is good evidence of potential impacts and their extent, stakeholder groups will feel no obligation to make contributions to or to assist with the process of resolving any environmental issues.

Assessments of the impacts of established ornamental species are also needed to feed information back into risk assessments for the importation of other ornamental fish into Australia. Coates and Ulaiwi (1995) attempted this when ground-truthing their exotic fish introduction risk assessment model in Papua New Guinea. Although such hind-casting approaches do not allow prediction of all the impacts of species, they might help validate or invalidate criteria used as part of other existing frameworks. This, in turn, will ensure that systems for assessing the risk of importation of ornamental fish species in Australia become more robust.

The literature on the risks of importation, introduction and establishment of alien fish species is littered with examples of authors who cite a lack of information on impacts. Koehn and Mackenzie (2004) are among the most recent Australian authors to note this as a problem. As a result, strategies for managing established exotic fish species in Australia are often based on predictions derived from risk assessment frameworks such as that developed by Arthington et al. (1999) and Bomford and Glover (2004). These usually rely on evidence of impacts from other countries and hence on potential impacts in Australia. For some species, predicted impacts are based on anecdotal evidence or on results from a limited number of studies. One of the problems with using this sort of generalised risk assessment is that it tends to either constitute a hind-casting exercise as when applied to well-established and well-studied species (such as carp, *Gambusia* and tilapia), or a best guess approach reliant on limited data that have not necessarily been critically assessed when applied to less well-studied species. It is the latter situation that is of most concern, because flawed predictions of impact can lead to either a lack of intervention where it is required, or to costs being incurred to tackle a resource management issue when it does not exist or is minimal.

For the 23 species of established ornamental fish covered in this report it is important to determine the extent of knowledge concerning their impacts and to evaluate the reliability of the information available. It is in everybody's interest to gain more clarity about the extent and nature of the impacts of established alien fish species. A statement provided in Arthington and Mackenzie (1997) sums up the current situation succinctly:

'there is a desperate need for hard data rather than anecdote and speculation'.

Although further investigations of the impacts of established ornamental fish species in Australia are no doubt warranted, surety to all stakeholders can only be gained if those studies are robust and provide useful information. Reviewing past studies provided us with an opportunity to identify the lessons learned so that the objectives of new studies are properly set and met when future monitoring and research is carried out. There have been relatively few critical reviews of the evidence of impacts of alien fish in Australia. Those by Weatherley and Lake (1967), Arthington (1991), Arthington and Blühdorn (1995), Arthington and Mackenzie (1997) and Clarke and Grosse (2000) examined the weight of evidence for impacts associated with several established alien fish species, but very few of the 23 established ornamental species were among these. Furthermore, even though the evidence of impacts was reviewed, none of these authors examined the way in which the data were collected or how the findings were interpreted and reported in any detail.

As part of this study, we were asked to critically review the current methodologies used to gather data on the impacts of exotic fish. Identifying the strengths and limitations of different impact assessment methodologies is a necessary precursor to reviewing evidence of impact. It also helps identify the most appropriate methods and approaches for future studies. This chapter therefore examines the methodological tools used to determine the impacts of exotic species on natural ecosystems in Australia as a precursor to assessing the evidence for impacts of ornamental fish. It includes an assessment of the way in which evidence of the impacts of exotic fish species is reported in the literature and it examines the 'levels of proof' required to establish acceptance of impacts. It overviews the various methods and types of approach that have been used to obtain evidence for or against different types of impact. It describes the main limitations that have been experienced in gathering evidence of impacts by the various methods and it provides guidance on the approaches and methods that can be used in the future to answer questions on the impact of ornamental fish introductions in Australia.

3.2 Establishing the 'burden of proof'

There are different types and levels of proof required in impact assessment and a knowledge of the 'burden of proof' (i.e., the type, amount and quality of information required by managers before they can accept that an impact is occurring and action is warranted) is rarely discussed in reviews of the impacts of exotic fish species. Nevertheless, it is an important issue to acknowledge and understand before undertaking any attempt to assess the weight of evidence for or against impacts.

The burden of proof tends to vary with stakeholder perspective. For example, conservation groups may require a low level of proof of impact and advocate a precautionary approach to exotic fish control principally because there is a lack of information and it is better to be 'safe than sorry'. There are some workers who start off with the premise that it is rare for the introduction of alien fish to have no consequences and that few introductions of ornamental fish have resulted in benefits to humans or the environment (e.g., De Iongh and Van Zon 1993; Welcomme, 1984 and Moyle, 1985 *cited in* Arthington 1991). Such sentiments are the ultimate in terms of a precautionary approach. However, they may lead to costly and unnecessary action if, in fact, there is no problem. Lodge and Shrader-Frechette (2003) suggest that we should strive to avoid approaches based on precautionary principles as much as possible.

In contrast, groups that have some responsibility for creating or managing the impact may require a much higher burden of proof based on peer-reviewed, scientifically-defensible, replicated studies. These can be costly, may take many years to complete and, in some cases, may be impossible to achieve. Some balance is therefore required in terms of the levels of proof of impacts required before policy initiatives are refined and implemented to tackle the issue of established ornamental fish.

Koehn (2004) argued for a balanced approach and indicated that it is better to assess the risk of invasive fish species based on qualitative information than to avoid considering risk at all, or to wait until semi-quantitative or quantitative data become available and provide a clearer indication of actual risk. Given the potential for lost opportunities to control alien fish at any point between their introduction and their establishment, this view is certainly valid and may help reduce the spread of some species. It is certainly a common sense approach to be advocated when fish introductions are limited to few sites and the potential for spread can be readily halted (i.e., site-led management). However, it may not be appropriate for species already well established in a wide range of locations and spreading. A more quantitative, wider-scale approach to impact assessment would be required to underpin the more expensive management required for these species (i.e., species-led management). The level of proof required for acceptance of an impact can therefore vary depending on both stakeholder perspectives and the geographic distribution (and spread) of the introduced species (i.e., site- versus species-led approaches).

The level of proof is also important in terms of type I and type II errors associated with data analysis and reporting. When assessing the impacts of established ornamental fish, committing type II errors (failing to detect an impact when it is present) could mean that resource managers overlook impacts and only realise that they are occurring much later, when the consequences are manifested and it is too late for remediation. On the other hand, if type I errors are committed (identifying an impact when it is not present), it could mean that money, time and resources are wasted on trying to remedy or mitigate impacts that are either non-existent or, at very least, too trivial to be considered ecologically significant. The level of proof required clearly needs to avoid such traps, especially type II errors as these could be much more expensive to fix and environmentally damaging in the long term.

The level of proof also depends on the type of impact assessment that is carried out and the scale of this. For example, aquaria and tank-based experiments (microcosms) or experiments in enclosures, limnocorals and artificial ponds (mesocosms) may reveal an impact that only occurs when exotic species are artificially constrained and this may not occur in the wild. For example, (Ling 2004) stated " *Gambusia is an aggressive little fish and cannibalistic, and commonly displays significant aggression towards other species and each other when confined in laboratory aquaria. Inter-specific competition is often directed to adult fish much larger than themselves. Such studies are often given as evidence that Gambusia may wreak havoc on wild populations. What is unclear is how closely these confined aquarium experiments mimic impact in natural habitats or larger scale, natural experimental systems"*. The level of proof of impact from such controlled experiments will clearly be less than that based on experiments using ponds or mesocosms let alone natural waters. In this respect, a high autumn mortality of *Galaxias gracilis* caused by fin-biting from *Gambusia* was recorded in a New Zealand lake indicating that tank observations of such behaviour can occur in the field (Rowe 2003).

Furthermore, the issue of the predictive power of impact assessments arises even when there is solid evidence of an impact in a natural environment. An impact from an exotic fish species may be demonstrated to a high level of proof within one or two natural waters, but this does not mean that every wild population will produce a similar impact. Impacts in the wild may vary greatly depending on site and time-specific factors. For example, gambusia may not have an impact on native fish in some lakes because the area of shallow littoral zone, where interactions occur, is small or not important for the other fish. Similarly, effects of gambusia on native fish in rivers may not occur until severe droughts or heavy abstractions reduce rivers flows, concentrating fish species within shallow pools. Ideally, impacts need to be scientifically demonstrated at a range of scales as well as over a wide geographic area before species-led control programmes are adopted. The level of proof required to demonstrate an impact therefore depends on the scale of the assessment experiment as well as on the generalisation of such results across a wider geographic region.

Because the level of proof required to demonstrate an impact depends on many factors and can vary greatly, it can provide a stumbling block for managers and a major issue between opposing stakeholder groups. Furthermore, economic factors may need to be considered by managers alongside proof of ecological impact before establishing control programmes. The 'burden of proof' of impact can therefore depend as much on socio-economic factors as it does on scientific ones. Because of this, a mutually-agreed consensus view from all key stakeholders, or a majority agreement among them as to what level of proof is acceptable, may be required. This recognises the fact that management decisions may be needed urgently and cannot always be based entirely on unequivocal scientific evidence. Reaching such agreements might be difficult. However, the aquarium industry will probably want a relatively high degree of proof of impact, particularly if the consequences of not having this proof will mean tighter regulations and negative publicity, or if funds are sought from the industry for investigations of perceived issues. In contrast, conservation groups are likely to opt for a much lower level of proof more in keeping with the precautionary principal. Ultimately, the 'burden of proof' required to trigger management action needs to be guided by both scientific principles and socio-economic considerations and determined *a priori* to avoid disputes.

3.3 Approaches to impact assessment

The review of the impact of exotic fish in Australia indicated that most studies can be broadly categorised into one of five major approaches which vary in the level of proof of impact provided. These are, from the lowest to highest level of information provided: (a) the existence of a breeding population of an introduced species in the wild, (b) a risk assessment approach based on one or more of niche theory, population dynamics and a history of invasiveness or pest status elsewhere, (c) a correlative or epidemiological approach based on the repeated observation of cause and effect patterns across both temporal and geographic scales, (d) a mechanistic approach based on the identification and experimental verification of all the causal links between and introduced species and its impacts, and (e) a triple bottom-line approach wherein ecological impacts are evaluated alongside social and economic ones. In the following section we discuss the values and limitations of each of these approaches as applied to studies on the impacts of exotic fish in Australian waters.

3.4 Presence of a breeding population

The establishment of an exotic fish population in a natural waterway can be regarded as an impact in its own right on the basis that it represents a deviation from naturalness (Kennard et al. 2005). Although the introduction of a new fish species may not have a detectable effect on the native fauna and flora or natural habitats, the existence of such a fish population requires resources for its production. In most cases, these will be drawn from sources formerly utilised by native species and in this sense the introduction has resulted in an impact. However, the salient point here is that it has not affected the natural ecosystem values considered important by society. Nonetheless, Arthington and Blühdorn (1995) are right in saying:

"All the established exotic fish species have one negative economic impact represented by the resources required to monitor, investigate, manage and, in a few isolated cases, exterminate them."

Pollard and Burchmore (1986) in their hypothetical outlook for what Australia's fish fauna might look like in 2000, reflected on the fact that their grandchildren would find a different fauna to the one that they had seen and experienced. In this context, the introduction of an exotic fish has changed the fish fauna from its natural state and in the eyes of these authors this constitutes an impact because it limits future knowledge and experience of natural aquatic ecosystems for future generations.

Many indigenous peoples in Australia regard the concept of naturalness as a fluid one, whereby the flora, fauna or ecological processes prevailing at any given time are what is natural. This view accepts that native fish communities will eventually be supplanted by alien species, but the extent to which this view is widespread or held by a majority of Australians is unknown. For some, the establishment of certain exotic species in some waters may be acceptable. For example, in their statement, *"I suppose we had to be grateful that at least some fish could still live in the more polluted of our rivers"* Pollard and Burchmore (1986) accept that exotic fish may be the only species that can survive in some highly degraded environments. In this sense, they fill a 'vacant niche' albeit a man-made rather than a natural one. Some non-ecologists may believe, rightly or wrongly, that invasions of new habitats may compensate for the extirpation pressures some species face in their natural habitat. For example, Botkin (2001) stated that: *"…biological invasions are natural and, more important, necessary for the persistence of life."* Furthermore, Flannery (2001) stated that: *"extinctions and* *invasions of biota characterised the Earth long before humans existed"*. However, Gurevitch and Padilla (2004) stated that: "*Most ecologists would not…regard the establishment of five new widespread alien species in a region as 'biotic compensation' for the extinction of five endemics."* Views on the role of exotic fish in 'new environments' or in 'unfilled niches' clearly differ and raise questions about how exotic fish species might change food webs and affect ecosystem stability and resilience. This more holistic approach to exotic fish impacts awaits further development of knowledge about the importance of ecosystem stability and resilience for ecosystem functioning and sustainability before it can be applied to impact assessments.

Although deviation from naturalness can be taken by some as an impact, most biologists rarely refer to this when reporting their findings with respect to a particular alien fish species. The implicit assumption is that it is not an impact that most of society is concerned with. However, this view is changing and incorporation of societal views on the issue of naturalness into future impact assessments may be required. Deviation from naturalness was addressed in relation to exotic fish by Arthington (1991) and is being increasingly adopted as basis for identifying the impacts of exotic species. Even if the presence of a breeding population of fish, without further evidence of impact, was to become a major concern in its own right, some benchmarking of the importance placed on naturalness would need to be carried out before it could be placed in a management, decision-making context.

3.5 Desk-top risk assessments

Another form of impact assessment occurs where authors use a combination of the maximum size of a fish species, its likely spatial distribution, any history of invasiveness, its known biological traits and reproductive potential, and its pest status in other countries to determine the potential for impact. This approach is based on the application of knowledge of the species gained elsewhere to predict potential impacts at locations where it is not yet present but may become established in. It is a precautionary approach, and has been likened to 'shooting first and asking questions later' (McDowall 2004). This approach can be justified when applied to potential new importations or to species with very limited existing geographical ranges in Australia because with no or few wild populations it is difficult to establish whether such species will pose a problem for the Australian fauna or not. However, this approach is of limited value when applied to predicting the impacts of species already well established in the wild. The impacts of widely established species can be determined by more scientifically robust approaches.

Pollard and Burchmore (1986) attempted to predict the future spread and impacts associated with alien fish species in Australia. Their predictions were necessarily couched in hypothetical terms and were acknowledged as representing a 'worst-case' scenario, but they did take into account other sources of environmental stress. An Australian example of a precautionary approach in reporting is provided by Blühdorn and Arthington (1990a) for tilapia (*O. mossambicus* and *O. mariae*). They identified niche separation and partitioning in the water column between tilapia and two other large native species (*Tandanus tandanus* and *Leiopotherapon unicolor*) with respect to use of food resources and foraging patterns and stated that *"no hard evidence is yet available on adverse effects (of tilapia) on native fishes"*. Despite this lack of evidence for impacts in Australian waters, they recommended in their Management Plan for Australia (with respect to tilapia) that all tilapiine species be declared noxious.

Some assessments of impact based on risk assessments assume that species judged likely to be good invaders will turn out to be pests. However, invasiveness and impacts are not always linked. For example, in an appraisal of the question "are invasive species a major cause of extinctions?" Gurevitch and Padilla (2004) stated that the *"link between species invasions and extinction of natives is widely accepted by scientists as well as conservationists, but available data supporting invasion as a cause of extinction are, in many cases, anecdotal, speculative and based upon limited observation"*. Invasiveness was originally applied to plants and terrestrial animals because of their rapid spread and damage to terrestrial ecosystems. It is an appropriate trait for defining the pest potential of species inhabiting terrestrial ecosystems, but is not necessarily a useful trait for defining the potential impacts of fish in aquatic ecosystems. The 'pestiness' of a freshwater fish species therefore needs to be based more on traits related to its impact on the native fauna or flora than on its invasive potential. The social and economic importance of any such impact will then be determined by its invasive potential.

Desk top assessments of the invasive potential of freshwater fish in Australia have been carried out by Arthington et al. (1999) and more recently by Bomford and Glover (2004). The former scored species on the basis of their previous success at invading other countries, their propagule pressure (a measure of total fish numbers imported and the probability of releases) and the extent of bioclimatic matching. The latter used a similar but greater number of scoring metrics (viz., climate match, overseas geographic range, history of establishment elsewhere, taxonomic group). Both assessments scored most of the ornamental fish species highly (i.e. high invasive potential) with the only marked differences occurring for Victoria Burton's haplochromis and the three spot cichlid (Table 3.1).

Table 3.1 Ratings of the invasive potential of ornamental fish species present in the wild in Australia

Common name	Scientific name	Potential risk of invasion	
		Arthington et al. (1999) method	Bomford & Glover (2004) method
Hybrid cichlid	*Labeotropheus/Pseudotropheus*	----------	-----------
Jewel cichlid	*Hemichromis bimaculatus*	Very high	Very high
Victoria Burton's haplochromis	*Haplochromis burtoni*	Very high	Low
Black mangrove cichlid	*Tilapia mariae*	Very high	High
Redbelly tilapia	*Tilapia zillii*	Very high	Very high
Mozambique tilapia	*Oreochromis mossambicus*	Very high	Extreme
Oscar	*Astronotus ocellatus*	Very high	Very high
Three-spot cichlid	*Cichlasoma trimaculatum*	Very high	Moderate
Jack Dempsey	*Cichlasoma octofasciatum*	Very high	High
Red devil	*Amphilophus labiatus*	High	High
Midas cichlid	*Amphilophus citrinellus*	Very high	
Convict cichlid	*Archocentrus nigrofasciatus*		High
Blue acara	*Aequidens pulcher*	Very high	Moderate
Green swordtail	*Xiphophorus hellerii*	Very high	Very high
Platy	*Xiphophorus maculatus*	Very high	Very high
Sailfin molly	*Poecilia latipinna*	Very high	Very high
Guppy	*Poecilia reticulata*	Very high	Extreme
Caudo	*Phalloceros caudimaculatus*	Very high	High
Three-spot gourami	*Trichogaster trichopterus*	Very high	Extreme
Oriental weatherloach	*Misgurnus anguillicaudatus*	Very high	High
Goldfish	*Carassius auratus*	Very high	Extreme
Rosy barb	*Puntius conchonius*	High	Very high
White cloud mountain minnow	*Tanichthys albonubes*	Moderate-high	High

Other desk top assessments have addressed the risk of disease importation and spread to native fish (AQIS 1999) but not the invasiveness or potential ecological impacts of these species. Clarke et al. (2000) assessed the environmental threats of various introduced pests in Australia, including goldfish, guppy and Mozambique tilapia, but did not address invasiveness or disease risk.

According to the report entitled Strategic Approach to the Management of Ornamental Fish in Australia published by the Department of Agriculture Fisheries & Forestry (DAFF) in late 2005 (DAFF 2005), the assessment of pest status is carried out independently across state jurisdictions, with both Queensland and Victoria undertaking recent reviews of their lists. A national noxious species list is now being considered and the criteria used for ascribing noxiousness are being examined in greater detail. All jurisdictions now base their assessments of noxiousness on the degree of 'pestiness', which can include aggressive behaviour, piscivorous diet, high fecundity, frequent spawning over a long life span, potential maximum size, broad habitat tolerances and morphological similarity to native species. The authors of the DAFF report felt that *"meeting one of these criteria alone was not sufficient to qualify a species as noxious; those species proposed for addition to a national list met many, if not all, of the criteria"*. We support this approach and it has clearly been adopted in various state jurisdictions for some species already registered as noxious but not yet present in Australia (e.g., Nile perch and walking catfish).

Risk assessment approaches are based largely on the informed judgement of experts in the field and most involve a ranking or scoring system for the various variables incorporated into them. So far we are unaware of any risk assessments that have used Bayesian probability to test or refine decision-making. Bayesian models are ideally suited for risk assessments especially where the number of primary variables is large and there are important secondary variables that may influence these. The application of Bayesian probability to risk assessments should therefore be considered in the future.

Although risk assessments may be useful tools for screening species for importation, they are less useful in predicting impacts once a species becomes established in the wild. Because of this, they should be viewed as a means of developing hypotheses about potential impacts which can then be tested using other approaches.

3.6 Correlative approaches

More detailed, quantitative assessments of the impacts of established ornamental fish may reveal that certain species have a greater or lesser extent of impact, or that the impacts are of a more defined nature than initially predicted. Such information is essential for natural resource managers to prioritise how funds for research, monitoring and control effort are allocated. But the identification of mechanisms linking an introduced species to impacts may be bypassed where similar impacts are recorded consistently and repeatedly across a number of sites and/or habitats (e.g. multiple lines of evidence point to the same conclusion). It would be erroneous and costly to assume that only deterministic, experimentally-verified identification of the links between cause and effect provides proof that a problem exists. Correlative and epidemiological approaches have also been successfully used to establish a link between a cause and its effect without full knowledge of the mechanisms involved. For example, cigarette smoking has now been strongly linked to lung cancer through epidemiological studies which do not reveal the precise mechanism of impact. Lodge and Shrader-Frechette (2003) stated that:

"…one often cannot easily determine what caused a cancer in a given case, so one must resort either to an empirically determined dose-response curve or a probabilistic model. Yet this failure to attain knowledge of cancer causation provides no grounds for denying either that the cancer rate, statistically speaking has been increasing…"

The repetition of patterns across a large number of samples is often used as an alternative to establishing causal links in both medicine and the social sciences where the mechanisms of cause and effect are highly complex and affected by numerous extraneous factors. For example, in clinical trials of a new drug, testing of applicability is often based on log-linear regression analysis of data from different patient treatment groups (Lodge and Shrader-Frechette 2003). Understanding the mechanisms involved in the success of particular drugs sometimes comes after such tests.

In the USA, an example of the correlative approach to impact assessment is provided by Moyle et al. (1986). They suggested that patterns they observed in relation to associations between carp, habitat parameters, native American fish communities and their prey were so widely repeated that explanations other than impacts of carp are unlikely.

More quantitative evidence is now required to support such statements. The relative abundance of native species as against exotic species and/or changes in species composition are the easiest parameters to measure and are most frequently monitored. Kennard et al. (2005) developed indices of ecosystem health which included the ratio of native to exotic species as well as the observed as against expected fish species composition for waterways in south east Queensland.

One of the more robust ways of testing impact hypotheses based on correlative evidence is the Before-After-Control-Impact (BACI), or beyond-BACI (multiple control location) approach, with the proviso that an appropriate level of replication relative to background variation has been used (Underwood 1991). These experimental designs are now highly regarded in Australia and are cited in a variety of national guidelines, including the ANZECC guidelines (ANZECC/ARMCANZ 2000) and the National Ocean Dredge Spoil Disposal Management Guidelines (E.A. 2002).

A range of novel univariate and multivariate statistical techniques is also now being increasingly used to establish negative associations between alien fish and native species. Some can even provide an indication of the amount of variation in native species composition explained by alien fish after that attributed to other factors has been taken into account. Examples of these types of study are provided by Gilliam et al. (1993) and Godhino & Fereira (1998). The use of a variety of statistical approaches as part of a single study can provide multiple lines of evidence for assessing the impact of alien fish on native fish, even where that evidence is based purely on correlation.

One of the problems with the correlative approach is that the causal links between any decline in native species and the introduction of alien fish are not established. Whereas changes in native species may coincide with the distribution of alien fish or the timing of their release and, in some cases the differences may appear marked, the impacts of unmeasured factors coinciding with these events cannot be ruled out. An example of this is the impact of degraded water quality on native species and the documented numerical dominance of established alien species over native species in such habitats. Bunn & Arthington (2002) also noted that impacts related to exotic fish may also correlate with changes in flow regulation. Although workers such as Kennard et al.

(2005) have been able to establish that the correlation between alien fish and poor water quality makes established alien fish potentially good indicators of ecosystem health, no workers have been able to prove categorically whether the decline in native fish in such regions is driven by direct interactions with alien fish, by a decline in water quality attributed to the activities of alien fish, or to anthropogenic degradation of water quality (i.e., unrelated to the activities on alien fish). We nearly always lack the pre-introduction data to be able to answer such questions and are caught in a 'chicken or egg' style dilemma. Even where there is a clear pattern of mutually exclusive distribution of alien and native fish, such as that observed for *Gambusia holbrooki* and the oxleyan pigmy perch *Nannoperca oxleyana* in Australia (Lloyd 1987, Lloyd & Walker 1986), there remains the possibility that the observed patterns are a result of unmeasured factors or processes unrelated to the actions of the alien fish.

Such problems need to be overcome by the application of manipulation experiments in which the prime cause of an impact is either reduced or eliminated to see if the impact is reversed, and *vice versa*. Recent Australian examples of this approach are the removal of trout from small streams by firstly, the creation of artificial barriers to their upstream movement and secondly by the removal of upstream populations (e.g., Lintermans 2000; Jackson et al. 2004). These measures resulted in galaxiid recolonisation of waters formerly occupied by the trout. Such 'management' experiments provide strong evidence that trout were the main cause of galaxiid decline in that particular system, and hence that other factors were not affecting the galaxiids.

A further limitation associated with the correlation-based approach is associating a lack of detectable change with a lack of impact (e.g., Kushlan 1986). This ignores the possibility that either the experimental design was not sufficient to detect changes over and above background variability, or that impacts may have resulted in consequences that simply haven't been measured. In most of these cases, workers may have only measured relative abundance, or species composition of fish, whereas there may have been more subtle impacts such as reduced condition or size, or increased susceptibility to pathogens that went undetected, but which could manifest themselves as changes in relative abundance or species composition at some later date. Absence of proof is not proof of absence. Therefore, in circumstances where significant impacts are not detected and the species in question has no identifiable beneficial value, it might be best not to assume that its effects are ecologically benign (Lodge & Shrader-Frechette 2003).

3.7 Mechanistic approaches

A good understanding of the impact mechanisms of exotic species can help resource managers both confirm a suspected cause and effect relationship as well as to identify the best methods of pest control or the mitigation of impacts. An understanding of impact processes is also essential for allowing resource managers to determine whether the impacts are acceptable or a major threat to the resources/values they are charged with protecting. For instance, managers can be expected to treat the mortality of a threatened species with far greater emphasis than a restriction in its range, a change in its distribution, or a shift in behaviour. Understanding mechanisms can also be useful in predicting the time it might take for impacts to manifest themselves as large-scale or serious long-term changes. Such information provides a timescale on which to base management responses to impacts. In ecological science, assembling all such pieces of information together is sometimes referred to as 'matching patterns with processes'. In Australia, there are growing calls for this approach to be used more in ecology and to move away from looking at patterns for their own sake (Constable 1999; Fairweather 1999). A greater focus on processes, rather than patterns, now needs to apply when assessing the impacts of established ornamental fish on Australia's waterways and native biota.

Perhaps the most common problem with measuring factors associated with impact mechanisms is the problem of scale. The demonstration of predation, competition and aggressive behaviour by alien fish toward native fish is often achieved only at the microcosm or mesocosm scale (Lodge et al. 1998; Ling 2004). Evidence often consists of visual observations that are repeated over time on a number of individuals, and occasionally, exclusion or manipulative experiments. Such experiments can provide good insights into linkages between impact mechanisms and impact consequences and may even aid in determining whether the impact mechanisms persist over a range of situations and life stages. For instance, researchers may wish to know whether fin-nipping behaviour exhibited by an alien fish towards a native fish only occurs during spawning or if food supply is limited, or whether it occurs regardless of such factors. They might also want to know what levels a shared resource must reach before competitive behaviour and/or exclusion becomes evident.

The results from micro- or mesocosm studies, or from artificial streams, are less easy to refute than findings based on correlation or association only, but their ability to adequately represent what might occur at larger scales under field conditions is often called into question (Ling 2004; Lodge et al. 1998). It is up to researchers to outline the limitations based on scale as clearly as possible when presenting their interpretation of the results. Replicating such experiments on 'large-scale, natural environments' may be theoretically possible (because of the presence of such environments), but it is rarely logistically possible because of either the cost or community or legislative resistance to manipulation of waterways on a larger scale for experimental purposes. As a result, evidence of impacts revealed in smaller-scale, micro- or mesocosms is often necessarily interpolated to larger spatial scales under natural conditions and the ground-truthing of this at larger scales under field conditions is not explored further due to the above constraints. Both are negative outcomes and workers in the field of exotic fish research and their management should be encouraged to try and persuade funding bodies to understand why it is advisable to extend laboratory-scale observations to observations in the field wherever possible.

Even where the impacts of alien fish are linked to the extinction of a native species, it may not be enough to simply establish the causal links. The question of whether or not these impacts are (or are likely to be) the primary cause of extinction also needs to be addressed. Very often, impacts of alien species are synergistic (additive) to those of other stressors. In the words of Gurevitch and Padilla (2004): *"Exotic species might be a primary cause for decline, a contributing factor for a species already in trouble, the final nail in the coffin or merely a bouquet at the funeral."* Establishing the relative contribution of exotic species to declines, extirpations or extinctions is a daunting task (Gurevitch & Padilla 2004) and closer examination of actual case histories is required to determine the role of exotic species. Gathering such information will help resource managers decide whether the removal of ornamental fish will prevent extinctions or extirpations from occurring, whether this is a waste of time, or whether their resources might be better served in mitigating the impacts of other stressors, or undertaking habitat restoration.

Aside from visual assessments of behaviour, and of manipulative and exclusion experiments, workers often assess diet and predator-prey responses when investigating the impact mechanisms and consequences of introduced fish (e.g., Arthington et al. 1990). Assessments of gut content can be used to either demonstrate overlapping trophic requirements, or to demonstrate that predation of native fish has occurred (and to what degree and also at what life stage). As dietary assessments relate to field conditions, they are of immense value for assessing actual and potential impacts on native fish. Where there are several co-occurring alien species, dietary analysis might indicate whether populations of some of those alien species are controlled by predation pressure from others (Ruiz et al. 1992). Such information could be critical, as targeting the predator species for control or eradication might result in a marked increase in the populations of the prey species. This, in turn, could result in unforeseen pressures on native species.

Apart from providing direct evidence of predation on native fish, diet studies can also provide inferences about potential inter-specific competition, particularly when the dietary preferences, feeding behaviour and feeding zones of both alien and native fish species are known and shown to overlap. For example, there is fairly good evidence that the diets of the ornate rainbow fish *Rhabdinocentrus ornatus* and *Gambusia holbrooki* overlap, intermittently at least (Arthington & Marshall 1999), so evidence of potential inter-specific competition between the two can be inferred. Note that evidence should still be sought that the dietary resources are in short supply before causality for any decline in *R. ornatus* can be assumed. At very least, inferences about abundance of prey items should be made based on visual observations or knowledge of the prey species' life history, as done by Arthington and Marshall (1999). Low dietary overlap might be interpreted as resource partitioning as a result of past competition (Connell 1980 *cited in* Arthington & Marshall 1999). Thus, observations of the degree of dietary overlap may need to be repeated over a range of sites and times before inter-specific competition can be eliminated as a potential contributor to low dietary overlap.

For all that dietary overlap studies can tell us with respect to the potential impacts of alien fish on native fish, Weatherley & Lake (1967) pointed out that the mere presence of prey items in the guts of alien fish species does not provide information about the severity of the effects of predation on the prey. Thus, dietary studies on alien fish often need to be accompanied by some assessment of the effects by which increased predation on prey can affect native fish. Townsend (2003) undertook such an exercise in his study on the impact of brown trout on native fish in New Zealand. He found that brown trout feeding behaviour in streams has probably changed the daily timing patterns of emergence of benthic invertebrates considered to be preferred prey items of native fish. Such changes in invertebrate prey behaviour induced by trout foraging could render the invertebrates less available to native fish and hence reduce native fish production. They also potentially reduce invertebrate production.

Indirect impacts of exotic fish on native species can be much harder to determine than direct impacts, especially given that workers need to be able to demonstrate the clear links between mechanisms and consequences at multiple levels and scales, and at the same time, avoid biases introduced by other co-varying factors. The main mechanisms by which introduced fish indirectly affect native species are through habitat removal or modification, including degradation in water quality. Such changes can, in turn, act as a stressor of native flora and fauna, producing a secondary impact. However, degradation in water quality resulting from other activities or events may be far more pronounced than that attributed to alien fish. Moreover, degraded water quality and the prevalence of alien fish seem to go hand in hand (Kennard et al. 2005), and this might reflect impacts, or that alien fish are better able to tolerate degraded water quality than native fish, or that alien fish are introduced to waters at the same time as the water quality becomes degraded from land-use changes in the catchment. The most notable examples of attempts to infer indirect impacts of established alien fish in Australia on native fish have been the assessments of carp impacts on turbidity and cyanobacterial bloom generation in Australia’s waterways and, to some extent, the impacts of tilapia (*Oreochromis mossambicus*) on turbidity. Such studies have more of a focus on water quality than on native fishes, although some authors acknowledge the potential negative effects of increased turbidity (and by virtue, sedimentation) on fish spawning habitat and the smothering of native fish eggs. In the case of the potential for carp to influence the prevalence of cyanobacterial blooms, other potential impact mechanism pathways have been put forward and include grazing of zooplankton that feed on these algae (top-down process) and the excretion of nutrients, many of which are derived from feeding on benthic flora and fauna that otherwise aren’t grazed by native fish (bottom-up process) (Gehrke and Harris 1994). It is an extremely complex process to determine which impact mechanism has more influence on the outcome in this case, let alone to gain quality evidence for even one of these mechanisms. However, studies like that carried out by Gehrke and Harris (1994), which are based on dietary studies of carp and native fish at a range of life stages and which provide conservative estimates of excretion rates for carp based on other benthophagous species, provide a weight of evidence for the probable relative (if not absolute) influence of zooplankton grazing and excretion on cyanobacterial blooms. In natural ecosystems there is typically a high degree of autocorrelation among key variables that can confound the ability or workers to provide unequivocal evidence of causative links where indirect impacts are concerned.

3.8 Triple bottom-line assessments

Another trend that emerged from our review of the literature on impacts of alien fish on native fish was the emerging importance of triple bottom-line style assessments. Admittedly, such studies are somewhat limited when it comes to ecological impact assessments in Australia (with the possible exception of activities that trigger the EPBC Act) and most of the articles we covered were from the scientific literature (i.e., mostly peer-reviewed journal papers). Triple bottom-line studies appear more common for alien fish species introduced to supplement fisheries stocks or to control aspects of the environment such as mosquito larvae abundance or the prevalence of aquatic weeds, than for introduced ornamentals. For these other species, the social benefits of their introductions are sometimes taken into account. Reports by Arthington & Blühdorn (1995) and Arthington & Mackenzie (1997) are Australian examples of studies that have attempted to present impacts of alien fish species in a triple bottom-line context, though unfortunately these reports do not cover many of the 23 species being investigated as part of this study. The NSW National Parks & Wildlife Service Threat Abatement Plan for *Gambusia holbrooki* (NPWS 2003) is another Australian example of a publication that covers economic and social aspects as well as outlining ecological impacts; though in this report reference to both economics and social aspects was made only in relation to control options.

The social and economic value of releasing aquarium fish into the wild is assumed to have little significance. Releasing such fish into the wild may make some people feel comfortable about not killing them. For others, the release may be deliberate and motivated by personal gain. Such positive social and economic outcomes are likely to be limited. A better understanding of the niches these species fill and of their interactions with ecological maintenance processes may ultimately reveal that, in some circumstances, some of the 23 established species perform ecological services not yet known to us. Some research into such questions is possibly warranted as such positive impacts need to be weighed against the negative ones.

3.9 Recommended approaches for ornamental fish in Australia

Ideally, in the case of ornamental fish established in the wild in Australia, the 'burden of proof' will be determined at a species level and agreed upon in advance by stakeholders before studies are approved and funded. Given that 23 species need to be assessed, a priority ranking is likely to be required and this will need to be based on a risk assessment incorporating both the potential ecological damage as well as the risk of spread.

For the ornamental fish species with only one or two known wild populations (see chapter 2), a site-led approach will be needed to determine whether eradication or control of spread is required. This is because eradication will be much less costly at the present stage of the incursion than should the species be spread more widely. In this respect a site-specific risk assessment is probably the best initial approach and this should focus on identifying potential vectors (natural and anthropogenic) that can potentially spread the species from the site(s) and, while focussing on containment, should also develop hypotheses of potential impacts which can then be tested at the site.

The correlative approach is unlikely to provide a viable impact assessment approach for the species with very limited distributions, and a mechanistic approach, based initially on microcosm and mesocosm experiments, would be more appropriate for these species. Should the results reveal an impact is possible, then this may require validation at the site before it can be accepted as a real risk.

There are huge and numerous knowledge gaps surrounding the impacts of the 23 listed ornamental fish species in Australia. As stated earlier, it is always ideal to combine elements of impact mechanism, impact consequence and impact manifestation when undertaking investigations into the ecological effects of alien fish species on native fish. Prior to the implementation of any such studies, conceptual models describing the interactions between these elements should be developed so that testable hypotheses can be more easily derived. It is only through testing hypotheses that researchers are able to provide certainty to resource managers with respect to the significance of environmental impacts. However, the need for this level of certainty will depend on whether resource managers, with the general consensus of key stakeholders, are prepared to opt for a precautionary approach to managing the impacts of alien fish (including established ornamentals) in Australia. If this is the case, then perhaps all that will be required is for researchers to be able to demonstrate that at least one of the main impact facets has occurred in association with a particular fish species or fish community[3]. The advantage of this may be that impact mitigation measures can be put into place earlier than if the full nature and ramifications of the impact(s) was determined; something that may be impossible to achieve. A disadvantage of this is

[3] i.e., if there is some evidence that supports the occurrence of impact mechanisms, impact consequences or impact manifestations

that a lack of understanding about the full nature of the impact may hamper the ability of resource managers to implement appropriate mitigation measures in the first place. Moreover, studies of the impacts of these established ornamentals in native Australian habitats will apply only to the habitats where they are established, and they may be uninformative or of only limited use across the range of habitats into which they can potentially be introduced and become invasive (i.e., assessments of existing impacts may be inadequate to make judgements on broader-scale impacts).

Eradication of some of these highly restricted populations without good evidence of impact might be considered prudent and precautionary if the cost of this is low and if eradication is feasible. Therefore an appraisal of the possibility for this would also be warranted for some species. This assumes that the cost of eradication at one or two sites will be less than the cost of efforts to determine whether impacts are occurring or not. This would involve a socio-economic and technical appraisal of the feasibility of eradication and is essentially a precautionary approach based on cost-benefit considerations.

For the species with multiple populations and widespread distributions, the cost of control and/or eradication can be expected to be high. For such species it is likely to be too late to consider large-scale eradication given current control technologies. A species-led approach to determine the existence or not of widespread problems is therefore warranted to identify whether management action (e.g. containment) is appropriate as wall as what management actions are needed. A correlative approach maybe feasible for some of these species, but this assumes good data have been collected on at least some environments before the introduction and will allow a **Before** and **After** comparison. It also assumes that there will be comparable environments where the species has not been introduced to that can act as **Control** (or reference) sites and so provide a **Control** and **Impact** comparison. If such sites exist, a full or partial BACI approach may be warranted.

If a BACI approach is feasible, the issue of what variables to measure and what hypotheses to test then arises. A more generalised, species-based risk assessment approach, coupled with micro and mesocosm experiments, is likely to be the best way of identifying these. But as the results of tank or mesocosm experiments might not adequately reflect what could happen on larger scales, or in the natural (ambient) environment where a much more complex set of processes and interactions is likely to occur, a mechanistic approach is then likely to be required to provide adequate proof of impact, or a lack of it. This will also need to address scale effects as a proven impact in one site won’t necessarily occur at all sites. Investigators should therefore be encouraged to find several ‘natural’ field experiments. Multiple sites for impact analysis also create opportunities to see whether impacts can be reduced if the introduced species density is reduced and *vice versa*. Townsend (2003) used such phenomena to great effect in his study. These situations may not exist but designs based around natural experimental scenarios are likely to provide the most reliable evidence of impact, so are well worth looking for.

A key consideration for any assessment of impacts of alien fish that focuses on changes in relative abundance or species composition in the wild is that monitoring of these parameters should incorporate appropriate capture techniques and levels of effort to ensure the information is as robust as possible. Pilot studies may need to be carried out for some of the 23 established ornamental fish species to determine the most appropriate gear types for sampling them and the levels of sampling effort required to reliably detect changes in their abundance. This development of sampling methods is also required for studies aimed at establishing their presence or absence in waters not yet sampled (see chapter 2).

4. Ecological impacts of aquarium fish

4.1 Introduction

This chapter provides information (where it is known) on the origins, geographical ranges, habitat preferences, physical tolerances, reproductive behaviour, feeding and diet of each of the listed established ornamental fish species covered in this report. It also reports on a variety of intrinsic characteristics of the listed species relevant to understanding invasiveness and it reviews known impacts by these species, both globally and in Australia.

A comprehensive literature search was carried out in order to undertake this review using both the common and scientific names of each species and including alternative names where these occur. The databases searched included: Aquatic Sciences and Fisheries Abstracts, Conference Papers and Abstracts, Environmental Sciences and Pollution Management, Web of Science, Fish and Fisheries Worldwide and Scopus. The citations of all references recovered were downloaded into ENDNOTE and an electronic version of this database will be made available to the public with this report as a downloadable file from the DEW website. The number of references per species is shown in Table 4.1 and the total number per species indicates the variability in coverage. The genera *Carassius* (goldfish), *Oreochromis* (tilapias) and *Poecilia* (sailfin molly and guppy) have received much more attention than the other species. In contrast, there were less than 10 references for the genera *Amphilophus, Haplochromis*, *Hemichromis, Labeotropheus, Tanichthys* and *Trichogaster*.

Table 4.1: Number of references retrieved from the literature searches. Databases searched included CSA (Aquatic Sciences and Fisheries Abstracts, Conference Papers and Abstracts, Environmental Sciences and Pollution Management), Web of Science (WOS), Fish and Fisheries Worldwide (FFW) and SCOPUS.

Genus of fish	Database searched				Total references found (excluding duplicates)
	CSA	WOS	FFW	SCOPUS	
Aequidens	6	0	6	0	12
Amphilophus	6	0	1	0	7
Archocentrus	25	0	1	0	26
Astronotus	16	0	11	3	30
Puntius/Barbus	20	0	6	0	26
Carassius	59	41	111	7	218
Cichlasoma	10	1	3	0	14
Haplochromis	0	0	0	0	0
Hemichromis	2	0	4	0	6
Labeotropheus	0	0	1	0	1
Misgurnus	10	6	5	0	21
Oreochromis	44	26	112	12	194
Phalloceros	10	1	2	0	13
Poecilia	45	16	44	6	111
Tanichthys	1	0	1	0	2
Tilapia	17	0	26	2	45
Trichogaster	1	0	6	1	8
Xiphophorus	6	1	9	0	16

The grey literature is much harder to access than scientific publications and to a large extent coverage of this depends on familiarity with the species and with researchers undertaking studies on them. State agencies were solicited for information on the listed species as were a number of key contacts in Australia. The information gathered from the grey literature proved to be more relevant to this review than that in the scientific literature, perhaps reflecting the fact that much of the work carried out on aquarium fish in Australia is funded by state agencies and therefore only reported in the grey literature.

Overall, the literature on many of the species is inadequate to provide comprehensive coverage of the topics listed above. Moreover, there is a substantial likelihood that what literature is available is not based on rigorous scientific studies. Thus, we have also utilised information in web-based databases (e.g., FishBase, the U.S. Geological Service Fact Sheets on exotic species in North America (USGS), and the Global Invasive Species Programme database on invasive fish species (GISP)) to derive information on the species reviewed. For example, Fishbase (Froese & Pauly 2006) lists the countries to which a species has been introduced and indicates whether adverse ecological impacts have been recorded in the scientific literature for any of these introductions. It does not provide a critical revue of such reports and it should be noted that much of the information in these databases is often derived from sources that are themselves derivative and without explicit data support. For example, FishBase[4] is an international 'on-line' compendium of information on virtually all known fishes. One of the key sources of information in FishBase is a series of books known as the Baensch aquarium atlases (Riehl and Baensch, 1991; Baensch and Riehl, 1993). These are, themselves, derivative and furthermore provide no documentation of the sources of their information. As the conclusions in the Baensch volumes cannot be verified, there is a danger of being caught up in a 'game of Chinese whispers'. This caveat needs to be remembered when reading this chapter.

4.2 Species assessments

(a) Hybrid cichlid (*Labeotropheus/Pseudotropheus* cross)

No information was available for this hybrid, but we were able to find details in FishBase for a species of *Labeotropheus* (*Labeotropheus fuelleborni* –Mbuna cichlid) that may have relevance for this hybrid cichlid (albeit that hybrids can have a different biology from the parent species).

[4] FishBase is a database that presents information that has been summarised from a variety of reports. The authors of these reports are not cited in the conventional scientific manner in FishBase. Only a bibliography is provided. Thus wherever we refer to FishBase, it should be assumed that information has come from a variety of sources.

Indigenous range: Tropical, latitudes 9-15°S (corresponding with ambient water temperatures of 22-25°C). Native to Africa, in Lake Malawi, including offshore islands (FishBase).

Introduced range: None reported.

Maximum size: Males reach 30.0 cm SL, but female is much smaller, c. 9 cm SL (FishBase).

Habitat preferences: Lentic systems similar to Lake Malawi with hard, rocky substrates. Often found feeding in shallow waters between 1 and 6 m depth (FishBase). Based on information about the diet and feeding behaviour of *L. fuelleborni*, rocky substrates may be required as a feeding habitat.

Environmental tolerances: *Labeotropheus fuelleborni* occurs only in fresh water and prefers pH of 7.5-8.5 (FishBase). The fact that it is endemic to Lake Malawi may also suggest a narrow environmental range.

Behaviour: Usually feeds alone, or in small groups with other species, but can form feeding schools of several hundred fish which 'raid' the feeding grounds of other species (FishBase). If the hybrid cichlid exhibits similar behaviour, this could represent either a form of inter-specific competition or a form of habitat modification that could affect Australian native fish. Exhibits territoriality over food supply growing on rocky substrates and can be aggressive.

Reproduction: Not known.

Generation time: Minimum population doubling time 1.4 - 4.4 years, which FishBase equates with medium level resilience. The hybrid cichlid generation time may differ from this.

Diet: Has a relatively broad diet, and feeds on everything from algae and aufwuchs cover on rocks (its main diet), to crustaceans, insects and plant matter.

Likelihood of natural dispersal: Uncertain. Invasiveness rated as moderate to high by Arthington et al. (1999).

Risk of human spread: Depends on popularity amongst aquarists and their behaviour. The volume of fish sold in Australia is relatively low (Table 7.1).

Impacts overseas: FishBase regards *L. fuelleborni* as harmless, but that is not to say that the hybrid cichlid is as well. The aggressive territorial behaviour exhibited by *L. fuelleborni* could be a potential impact mechanism for this hybrid, as could intra-specific competition for algal food resources, were the hybrid cichlid forms schools that raid the feeding grounds of native species. This behaviour could indicate potential for competition over food with some Australian native fish.

Impact in Australia: Unknown. None reported to date. Current known distribution in Australia limited to Hazelwood Power Station ponds.

(b) Jewel cichlid (*Hemichromis bimaculatus*)

Indigenous range: Tropical, latitudes 4-11°N (corresponding with ambient water temperatures of 21-23°C). Native to Africa and considered to be widely distributed in West Africa. Specific locations where found include coastal basins from Southern Guinea to Central Liberia, where it is associated with forested biotopes, Côte d'Ivoire and Ghana, coastal basins of Cameroon, the Democratic Republic of the Congo and Nile basins, as well as Gambia and Senegal (FishBase).

Introduced range: Canada, Hawaii, Philippines, United States (FishBase).

Maximum size: Reportedly up to c. 14 cm SL and 10.0 g in weight in the wild (FishBase).

Habitat preferences: Occurs in mud- and sand-bottomed canals some distance inland from the coast, associated with areas of intact or recently disturbed forest cover (FishBase).

Environmental tolerances: Broad salinity tolerance but a narrow pH tolerance range of 6.5 to 7.5 (FishBase). Its association with forested biotopes within its native range may be an indication that this species is less tolerant of highly disturbed or polluted conditions compared to some of the other ornamental fish species established in

Australia, but this would need to be tested. Shafland & Pestrak (1982) reported a lower lethal temperature of 9.5˚C for this species so it has a potentially broad temperature range over which it can survive. This could allow it to occur in many parts of Australia but temperature requirements for spawning and egg survival are unknown and these may restrict its distribution.

Behaviour: Considered aggressive (FishBase), but no information given as to whether this is at all times or only at certain times (e.g., during spawning). Riehl and Baensch (1991) class it as “territorial and peaceful except during spawning, then the species is aggressive and intolerant”. Is also reported to disturb sediments when digging nests.

Reproduction: This species is potamodromous (FishBase), so probably requires a migration for spawning. This might limit its ability to establish where introduced to closed systems. Sediment is disturbed to create nests. Both sexes become bright red when mating (Midgalski & Fichter 1977) and so may be more vulnerable to other predators at this time. Spawning is probably stimulated by factors associated with rainfall (e.g. water level rise, increased water movement) and eggs are attached to hard objects (Siddarth 2005).

Generation time: Minimum population doubling time less than 15 months, which FishBase equates with a high degree of resilience. Believed to reach maturity in less than 1 year and to be capable of multiple spawnings in a given year.

Diet: Diet quite broad, eating everything from algae and aufwuchs to macrophytes, crustaceans and insects.

Likelihood of natural dispersal: Tolerates brackish water conditions, so unassisted range expansion in open systems is not likely to be restricted by reaches of brackish water connecting freshwater systems, such as harbours or bays into which several rivers flow. Invasiveness rated as very high by both Arthington et al. (1999) and Bomford & Glover (2004).

Risk of human spread: Relates to its popularity as an aquarium fish and the behaviour of aquarists. This species is rated as of medium importance to the aquarium trade in Australia and relatively low volumes are sold (Table 7.1). Risk of spread is therefore relatively low.

Impacts overseas: FishBase regards this species as harmless.

Impacts in Australia: Unknown. None reported to date. Currently known from only two sites in Australia (Rapid Creek near Darwin and Ross River in northern Queensland).

(c) Victoria Burton's haplochromis (*Haplochromis burtoni*)

Indigenous range: Tropical, latitudes 3-9°S (corresponding to ambient water temperatures of 20-25°C). Native to Africa, specifically Lake Tanganyika and associated rivers and Lake Kivu (FishBase).

Introduced range: Not reported elsewhere.

Maximum size: Grows to 15.0 cm SL (FishBase).

Habitat preferences: Inhabits slow streams, river mouths and shallow parts of lakes near the confluence with rivers (FishBase).

Environmental tolerances: Some members of the genus *Haplochromis* can tolerate low oxygen conditions, whereas others prefer oxygen-rich water (Galis & Smit 1979 *cited in* Obordo & Chapman 1997). Where Victoria Burton's haplochromis sits in relation to low oxygen tolerance is yet to be determined.

Behaviour: Nothing reported.

Reproduction: Little reported; eggs deposited on rocky substrates, young brooded in mouth of adult.

Generation time: Minimum population doubling time is less than 15 months, which indicates a high degree of resilience (FishBase).

Diet: Dietary preferences are not reported in FishBase, or elsewhere.

Likelihood of natural dispersal: Unknown. Invasiveness rated as very high by Arthington et al. (1999) but low by Bomford & Glover (2004). This difference reflects differences in the metrics used by these two methods to rate invasiveness.

Risk of human spread: Usual risks associated with keeping the species in captivity. The importance of this species to the aquarium industry is relatively low (Table 7.1) so the risk of spread is also low.

Impact overseas: FishBase regards this species as harmless.

Impacts in Australia: Unknown. None reported to date. Currently known from only two sites in Australia (Ross River in northern Queensland and Hinze dam in southeastern Queensland).

(d) Black mangrove cichlid (*Tilapia mariae*)

Indigenous range: Tropical, latitudes 2-9°N (corresponding to ambient temperatures of 20 – 25°C). Native to Africa, specifically the coastal lowlands and lagoons of the Tabou River (Côte d'Ivoire), southwest Ghana and southeast Benin to the Kribi River in Cameroon (FishBase).

Introduced range: Florida (FishBase, USGS).

Maximum size: To c. 40 cm TL; maximum published weight 1,360 g (FishBase).

Habitat preferences: Occurs in warm springs and in mud-bottomed to sand-bottomed canals (FishBase). Cadwallader et al. (1980) indicated that it is found in both still and flowing waters, over both rock and mud substrates, and below overhanging banks as well as where there is no cover (i.e., a wide range of habitats).

Environmental tolerances: Reasonably broad, given that it can occur in brackish as well as fresh water, and in waters of pH 6–8 (FishBase). Shafland & Pestrak (1982) found that the lower lethal temperature for this species was 11.2˚C so it has a relatively broad temperature range for survival. This could allow it to occur in many parts of Australia but the minimum temperatures for spawning and egg survival are unknown and if these are higher they may restrict it to warmer latitudes than its temperature tolerances suggest.

Behaviour: There are reports that Florida populations of this species are aggressive towards other species of fish (Courtenay & Hensley 1979 *cited in* Arthington & Blühdorn 1995). This aggression may be a potential risk for some Australian native fish. Regarded as territorial and pugnacious by Riehl & Baensch (1991).

Reproduction: Sterba (1966) suggests 150-200 offspring per spawning, but FishBase and Riehl & Baensch (1991) state around 2,000 eggs. Sterba (1966) indicated that this species requires well-cleaned stones on which to lay eggs but McKay (1984) indicated that submerged logs and aquatic plants are also utilised as a spawning substrate.

Generation time: Minimum population doubling time given as 1.4 - 4.4 years, giving this species a medium level of resilience. Time to reach maturity is less than 1 year (FishBase).

Diet: Demersal in habitat, and so unlikely to feed on prey found in the middle and upper reaches of the water column (FishBase). Is primarily herbivorous (Riehl & Baensch 1991). All tilapia are voracious feeders and many prefer algae and soft-leaved plants (Sterba 1966). This feeding behaviour may potentially result in habitat modification with indirect effects on water quality and hence the native biota.

Likelihood of natural dispersal: Ability to tolerate brackish conditions would allow it to move between waterways in open systems connected by brackish water reaches more easily than many of the other established ornamental fish. Relatively large size as adults would possibly afford it a greater degree of mobility than many other ornamental fish species, making it less likely that its distribution would be restricted by high flow conditions. Invasiveness rated as very high by Arthington et al. (1999) and high by Bomford & Glover (2004).

Risk of human spread: As above, but the risk of spread from one catchment to another will depend on the behaviour of the public and their knowledge of the potential impacts of making releases into natural waters. The volume of fish sold in Australia is relatively low (Table 7.1) so the risk of spread is also low.

Impacts overseas: FishBase regards this species as harmless, but Courtenay & Hensley (1979) indicated that it is now dominant in many canals in south east Florida, so has the potential to affect other fish. It is also dominant in Roger's Spring (Nevada) where there are concerns over its effect on native fish because of competition for food (Courtenay & Deacon 1983).

Impacts in Australia: Classified as a noxious species in Queensland under the Fisheries Act (Arthington & Blühdorn 1995) but this is based more on its potential for impact than on measurement of actual effects. Rated as a high-risk species under the Bomford & Glover (2004) risk assessment model. All *Tilapia* species are included in the Department of Agriculture, Fisheries and Forestry (DAFF) proposed noxious fish list (DAFF 2005) as well as in the Queensland noxious fish list (DPIQ 2000) indicating serious concerns about their potential to cause ecological impacts. However, none have been reported to date for this species. Currently well established in lowland waters near the coast in northern Queensland and also present in Lake Burley Griffin (Canberra) and the Latrobe River (Victoria).

(e) Redbelly tilapia (*Tilapia zillii*)

Indigenous range: Tropical to sub-tropical, 35°N and 10°S (equating to temperatures of 7- 43°C). Native to Africa and Eurasia, specifically, the Senegal River in the Niger-Benue system of South Morocco, the Sassandra and Bandama Rivers in the Volta system of the Côte d'Ivoire, the Ubangi-Uele-Ituri Rivers in the Chad-Shari system of the Democratic Republic of the Congo, the Lake Mobutu in the Nile system and in Lake Turkana and the Jordan system (FishBase).

Introduced range: Widespread; including Ethiopia, Guam, Hawaii, Japan, Madagascar, Philippines, Singapore, Syria, Tanzania, United States (FishBase).

Maximum size: According to published figures, reaches 40.0 cm SL and 300 g. Can reportedly live for 7 years (FishBase).

Habitat preferences: Prefers shallow, vegetated areas. Fry are common in marginal vegetation and juveniles are found in the seasonal floodplain. Typically found in waters up to 1m in depth (FishBase).

Environmental tolerances: Very broad in terms of survival, but probably less so in terms of requirements for spawning (see above). Can tolerate brackish as well as freshwater conditions; wide water temperature ranges (see indigenous range section above); pH 6-9.

Behaviour: Mainly solitary but occasionally forms schools (FishBase). Sterba (1966) classed it as pugnacious and stated that it exhibits strong brood care behaviour. Such behaviour might represent a potential impact mechanism as far as Australian native fish are concerned, but this remains to be tested formally. Territorial and pugnacious, and is a substrate burrower (Riehl & Baensch 1991).

Reproduction: Not a mouth-brooder (FishBase; Sterba 1966). Can produce up to 1000 eggs per spawning (Sterba 1966). Young are tended by adults. Is potamodromous, so a movement to spawning habitats is probably required. Is also a substrate spawner, requiring clean stones to lay eggs on (Sterba 1966). Larvae of this species develop in close association with the substrate (FishBase). There may, therefore, be opportunities for control involving access to suitable spawning habitats.

Generation time: Minimum population doubling time 1.4 - 4.4 years, giving this species a medium level of resilience; time to reach breeding maturity between 2-3 years (FishBase).

Diet: Relatively restricted compared to other tilapiine species establish in Australia. Generally herbivorous, feeding on aquatic plants and epiphyton, but will eat invertebrates – worms, insects, zooplankton, shrimps, gastropods (FishBase). Sterba

(1966) stated that it feeds eagerly on plants; this may result in alteration of habitats and/or deplete preferred foods of native fish species in Australia.

Likelihood of natural dispersal: Ability to tolerate brackish conditions would allow it to move between waterways in open systems connected by brackish reaches. Its relatively large size would possibly afford greater mobility, reducing the likelihood that upstream distribution would be restricted by higher water velocities or small rapids and falls. Invasiveness rated as very high by both Arthington et al. (1999) and Bomford & Glover (2004).

Risk of human spread: Is a prized commercial aquaculture and fisheries species in other countries (FishBase). Potential to reach a relatively large size as adults could result in this species being targeted and dispersed further by coarse fish anglers. It was introduced widely in the southern USA for control of aquatic plants as well as for control of mosquito and chironomid larvae (USGS). Could also be dispersed through use as live bait. However, the stunted growth of Australian populations of this species in South Western Australia (Blühdorn & Arthington 1990b) and their mainly herbivorous diet may reduce their suitability as an angling species.

Impacts overseas: FishBase regards this species as a potential pest and states that several countries report adverse ecological impacts after introduction. No further details are provided in FishBase about the mechanisms or manifestations of those impacts. Sterba (1966) stated that the species is a great 'digger' implying a potential impact on substrate created when it excavates nests for spawning. However, vegetation is also uprooted as a consequence of feeding on vegetation and while foraging for invertebrates. A potential impact mechanism associated with this species is therefore modification of aquatic substrates. It has been implicated in the decline of the desert pupfish (*Cyprinodon macularius*), and is reported to be highly aggressive in California waters (USGS). It poses a threat to fish dependent on aquatic plants. Crutchfield (1995) reported that all macrophytes were removed from a Wyoming reservoir 2 years after its introduction and several native fish species subsequently declined. However, the redbellied tilapia thrived because if could feed on other foods.

Impacts in Australia: All *Tilapia* species are included in the Department of Agriculture, Fisheries and Forestry proposed noxious fish list (DAFF 2005) as well as in the Queensland noxious fish list (DPIQ 2000) indicating concerns over their impact. However, no impacts of this species have been reported in Australia to date. Currently known to be present in only one location in Australia (Chapman River, Western Australia).

(f) Mozambique tilapia (*Oreochromis mossambicus*)

Indigenous range: Tropical, at 13°S and 35°S (equating to temperatures of 8 – 42°C). Native to Africa, specifically the lower Zambezi, the lower Shiré and coastal plains from Zambezi delta to Algoa Bay and southwards to the Brak River in the eastern Cape and in the Transvaal in the Limpopo system (FishBase).

Introduced range: Most commonly used species in aquaculture and once also stocked for aquatic plant and insect control in the USA. Now widely established beyond its natural range, in the countries of Caribbean, Czech Republic, Central America, Fiji, French Polynesia, Hawaii, Hong Kong, India, Kiribati, Nauru, Niue, Seychelles, Singapore, Solomon Islands, Thailand, Vietnam; and widely in the southern States of the USA (FishBase).

Maximum size: Maximum published size 39.0 cm SL or 1,130 g; individuals can live for up to 11 years (FishBase).

Habitat preferences: Prefers well-vegetated shallow waters, usually still, or gently flowing waters, most common in blind estuaries and coastal lakes, but also found in warm, weedy pools of sluggish streams, canals, and ponds. Found in waters up to 20 m in depth (FishBase). Normally lives in brackish water and grows more slowly in fresh water (Midgalski & Fichter 1977), so could proliferate in the estuarine regions of rivers and/or in inland brackish lakes.

Environmental tolerances: Has broad temperature and salinity tolerances. FishBase states that this species can be reared under hyper-saline conditions; also that *O. mossambicus* exhibits considerable plasticity in feeding habits and reproductive biology, suggesting that it can modify these according to the prevailing environmental conditions. Arthington (1986) reviewed the international literature on this species and reported a salinity range of 0-120 g/L after acclimation, with breeding between 0-49 g/L. The pH range tolerated was 4-11 and a lower temperature tolerance of 8-10ºC. Shafland & Pestrak (1982) found that the lower lethal temperature for this species was 9.5˚C. Its relatively broad temperature range, reported as 8-42ºC in McKenzie et al. (2000), could allow this species to occur in many parts of Australia, although Brisbane is considered by some to be its likely southern-most limit. It will stop feeding at temperatures below 8-10°C according to Clarke et al. (2000) and below 15ºC according to Philippart and Ruwet (1982) as cited in Arthington (1986).[5] Bruton and Bolt (1975) were cited in Arthington (1986) as reporting a temperature range of 20-24ºC for breeding but the temperature requirements for egg survival are unknown. Arthington & Mitchell (1986) reported that *O. mossambicus* has a capacity to tolerate

[5] Differences in temperature thresholds for a species often occur in the literature because temperature tolerances can vary with fish size and the size of fish tested is sometimes not reported.

low oxygen conditions and can survive levels as low as 1 mg L-1. Furthermore, it can tolerate high turbidity, allowing it to live in silty lagoons or degraded waterways often found in association with urban areas (Arthington & Mitchell 1986). Of more concern, perhaps, is its reported ability to bury itself in the moist upper layers of sediment in sandy streams (up to 3 m deep) as a drought survival mechanism (Donnelly 1978 *cited in* Arthington & Blühdorn 1995; Clarke et al. 2000). If established populations in Australia are able to do the same, this species could survive in inland systems that are ephemeral in terms of surface water flow.

Behaviour: Usually solitary, but may form schools (FishBase). Males require large territories and defend them against each intruder with aggressive behaviour (Sterba 1966). Such behaviour may present a potential impact for Australian native fish.

Reproduction: FishBase reports reproductive outputs of between 150-200 offspring per spawning. Arthington (1986) reported fecundities of 438-490 per 100g of female fish for two reservoirs near Brisbane. However, Arthington & Mitchell (1986) stated that a 100 g female can produce up to 600 eggs, though brood sizes are much smaller, and around 250. Although Clarke et al. (2000) suggested production of up to 1,700 offspring per spawning, fecundity can be expected to increase with fish size. Whereas large females may well produce 1,000-2,000 eggs, fecundity in stunted populations, which are the norm, is likely to be much less. The males excavate a shallow, basin-shaped depression where eggs are laid and these are picked up by the female and fry hatch in her mouth after 3-5 days. Fry are protected (in the mouth of females) for around three weeks (Clarke et al. 2000). Males often mate with several females over a short period of time (Arthington & Cadwallader 1996), giving this species a 'head start' in terms of reproductive output. Water temperatures above 23°C are required to induce spawning (Clarke et al. 2000), suggesting that this species could not undertake spawning in many parts of Australia during winter, though spawning could be achieved almost year-round in tropical regions. In Queensland, the breeding season extends from September-October and March-May (Arthington & Mitchell 1986; McKenzie et al. 2000).

Generation time: Minimum population doubling time 1.4 - 4.4 years, which, equates to a medium level of resilience (FishBase). Time to reach breeding maturation is less than 1 year (FishBase). Arthington & Mitchell (1986) regard this species as reproductively precocious and capable of reaching breeding maturity in 3-4 months. New broods can be produced every 4-6 weeks (Arthington & Mitchell 1986).

Diet: Very broad and plastic; a benthopelagic, omnivorous species, which, according to FishBase, feeds on almost anything from algae to insects (including terrestrial insecta). Exhibits ontogenetic shifts in diet, however, so dietary requirements for different life stages are probably more specific than the above indicates. Juveniles are carnivorous, but adults tend to be herbivorous (FishBase). Adults become increasingly herbivorous, preferring algae and soft-leaved aquatic plants (Sterba 1966). Sterba (1966) states that "all *Tilapia* are voracious feeders", which means that this species may have the potential to rapidly remove aquatic plant habitat and indirectly affect both native flora and hence fauna. This remains to be examined. Dietary studies in Australia indicate that feeding modes may differ depending on location. For example juveniles in the Chapman River, Western Australia were detritivores whereas in the Gascoyne they were insectivores (ASFB 2003b).

Likelihood of natural dispersal: Ability to tolerate brackish conditions would allow this species to move between waterways in open systems connected by brackish reaches, especially in harbours and river estuaries, and to do so more easily than many of the other established ornamental fish. The relatively large size of adults would possibly afford them a greater degree of mobility than many other ornamental fish species, making it less likely their distribution would be restricted by high flow conditions. Invasiveness rated as very high by Arthington et al. (1999) and extreme by Bomford & Glover (2004).

Risk of human spread: Is a highly prized aquaculture species in many parts of the world so could be spread because of its aquaculture potential. It was once stocked in the USA for plant and insect control as well as a sports fish and a food source. It may therefore be targeted and dispersed by coarse fish anglers, which Norm Milward (formerly senior lecturer in zoology and aquaculture at James Cook University) raised as a concern (Arthington & Blühdorn 1995). It has already been introduced deliberately to ornamental ponds at Port Douglas (A. Arthington, pers. comm.).

Impact overseas: FishBase reports this species as a potential pest and it has established itself in the wild in many countries where adverse ecological impacts have been reported. Competition with local species for resources is one perceived impact mechanism (FishBase). In Hawaii, this species is suspected of reducing the population of a valuable mullet species, *Mugil cephalus*, by competing aggressively for the same food resources – detritus and soft algae (Randall 1987 *cited in* Casal et al. 1999). Populations of mullet also decreased after this species was introduced to Kiribati, as did other benthophagous species, including bonefish and milkfish (Eldredge 1994 *cited in* Casal et al. 1999). Attempts to eradicate the Kiribati populations proved unsuccessful (Lobel 1980 *cited in* Casal et al. 1999). A similar scenario occurred in Nauru in terms of perceived impacts of *O. mossambicus* on milkfish and failure to eradicate established populations of this species (Nelson & Eldredge 1991; Eldredge 1994 *cited in* Casal et al. 1999). Swift et al. (1993) considered this species a major factor in the decline of the desert pupfish (*Cyprinodon macularius*) in the Salton Sea area. The introduction of this tilapia has even been blamed for the extinction of two duck species (*Anas superciliosa* and *A. gibberifrons*) in the Solomon Islands (Nelson & Eldredge 1991; Eldredge 1994 *cited in* Casal et al. 1999). While the evidence for these impacts is mainly circumstantial the number of negative reports indicates that there is an urgent need for investigation of the potential impacts of this species on native Australian fish, especially impacts on native, benthivorous species such as *Mugil cephalus*. Canonico et al. (2005) reviewed the effects of tilapias throughout the world and found correlative evidence for the decline of native fish in Madagascar, Nicaragua and Mexico following the introduction of a number of tilapia species, including *O. massambicus*. Unfortunately the role of *O. massambicus* in Madagascar could not be determined because of the number of other exotic fish present. In the Philippines, both *O. niloticus* and *O. massambicus* were introduced to enhance fisheries. Although *O. niloticus* is thought to have played a role in the decline of native fish in lakes Lanao and Buhi, the impact of *O. mossambicus* is not clear (Canonico et al. 2005). However, in Mexico, *O. mossambicus* became the dominant species in Lake Chichincanab and competed with a native cyprinodont fish for habitat resulting in its decline and threatening extinction.

Impacts in Australia: Arthington & Mitchell (1986) regard this species as a major threat, based mainly on its invasiveness and the potential for its dispersal to be heavily human-assisted because of its fisheries and aquaculture values. These authors recommended further studies to investigate the detrimental ecological effects of this species in Australia. Arthington & Cadwallader (1996) classed it as a pest in Australia. Arthington & Milton (1986) stated that the most likely impact mechanisms for *O. mossambicus* on native Australian fish was either competition for food or predation, while the manifestation of these impacts would most likely be the displacement of native fish species. Arthington & Blühdorn (1994, 1996) and Arthington et al. (1994) found no direct evidence of environmental impacts on the native fish fauna in a reservoir in the Brisbane area and Clarke et al. (2000) reported that although a few studies had been carried out on the impacts of this species in disturbed conditions within modified waterways (e.g., water reservoirs), no significant adverse impacts were observed. However, when drought conditions and low river flows resulted in the formation of pools in the Gascoyne River (western Australia), the Murchison River hardyhead (*Craterocephalus cuneiceps*) rarely occurred where the Mozambique tilapia was present (ASFB 2003b). Male tilapia were observed to be aggressive to native fish, and much of the substrate was covered by their nests. Although this suggests displacement of native fish by the tilapia, mechanistic studies (e.g., manipulations of tilapia abundance in such pools) are now needed to confirm this. McKenzie et al. (2000) indicated that there was anecdotal evidence that macrophyte beds were disturbed during nest building. This species is rated an extremely high-risk species under the Bomford & Glover (2004) risk assessment model which assesses the potential of a species to spread and become prolific (i.e., its invasiveness). Temperature tolerances suggest establishment is likely to be mostly northern. The Mosambique tilapia has been declared a noxious species in Queensland, the Northern Territory, New South Wales and Victoria with heavy penalties for transport and possession (Arthington & Blühdorn 1995). Possession is prohibited in South Australia and commercial utilisation of this species is not permitted in Western Australia (Arthington & Blühdorn 1995). All *Oreochromis* species are included in Queensland's list of noxious fish (McKenzie et al. 2000) and in the Department of Agriculture Fisheries and Forestry proposed list of noxious fish for Australia (DAFF 2005). Such listing is clearly a precautionary approach as there is currently little solid evidence of impacts despite major concerns over this species. Currently known to occur in four widely separated locations in Queensland and two in Western Australia.

(g) Oscar (*Astronotus ocellatus*)

Indigenous range: Tropical (latitude 4°N-15°S, equating to a temperature range of 22-25°C). Native to South America: Orinoco and Amazon River basins in Peru, Columbia, Brazil. Also in French Guiana to north Paraguay (FishBase, USGS).

Introduced range: Guam, Puerto Rico, Singapore. Reported widely in the USA (Lee et al. 1980) and established in Florida and Hawaii (USGS, FishBase).

Maximum size: 40 cm TL, but known for its slow growth rates (FishBase).

Habitat preferences: Prefers quiet, shallow waters, primarily mud or sand-bottomed ponds or canals (FishBase).

Environmental tolerances: Has relatively broad temperature tolerances, based on the lower lethal temperature of 12.9˚C reported for this species by Shafland & Pestrak (1982). This could allow it to occur in many northern parts of Australia, but as for other species of cichlid, the minimum temperatures for spawning and egg survival may restrict it to warmer waters. Restricted to fresh water (FishBase) so is unlikely to disperse between sites where migration spanning brackish or marine water is required. Its pH range of between 6 and 8 (FishBase) indicates a tolerance of moderately acidic as well as basic waters. Is capable of surviving large fluctuations in oxygen levels and is reported to be highly hypoxia-tolerant. It is believed to cope with these conditions by reducing its metabolic rate (Muusze et al. 1998). The capacity to tolerate low oxygen conditions may allow this species to readily colonise degraded waterways often associated with urban areas.

Behaviour: Basically peaceful in captivity, except when spawning, but is known to eat other fish in the laboratory and this may extend to native fish in the wild in Australia. Is also known to bite other fish and put on aggressive fin displays when defending territory during spawning (Beeching 1997). Such behaviour is undertaken by both sexes, though males are inclined to display some attack behaviours (such as charging) more frequently than females. A laboratory study by Beeching (1992) found that this species can assess the size of other fish visually, determine how large they are in comparison to its own size, and use this information to establish the intensity of its aggressive response. He also found that smaller intruders are more likely to be targeted. This could mean that smaller native freshwater fish co-occurring with *A. ocellatus* are potentially more vulnerable to any intra-specific competition for space posed by this species. Is territorial but peaceful in aquaria but has a tendency to burrow into substrate (Riehl & Baensch 1991).

Reproduction: Fecundity 300-2000 progeny per spawning (FishBase). Is a substrate spawner (Beeching 1992), but few details given. Eggs deposited on rocks, hatch in 2-3- days; young are guarded.

Generation time: Minimum population doubling time is less than 15 months, giving this species a high resilience (FishBase).

Diet: Feeds on small fish, invertebrates, including crayfish, worms and insect larvae (FishBase). Ingested live goldfish and *Gambusia* in laboratory experiments (Beeching 1992; 1997) and may well exhibit some piscivorous tendencies towards Australian native fish in the wild.

Likelihood of natural dispersal: No explicit risks identified. Temperature tolerances suggest mostly northern. Invasiveness rated as very high by both Arthington et al. (1999) and Bomford & Glover (2004).

Risk of human spread: Highly prized food fish in South America (FishBase) and recognised as a sports-fish species in Florida (Kushlan 1986), so it could be introduced into new water bodies by coarse fish anglers, given the opportunity. In Australia, this species is important to the aquarium trade with the volume of fish sold being rated as medium (Table 7.1). The risk of spread by humans is therefore relatively high.

Impacts overseas: FishBase regards this species as harmless. It is now a substantial part of the recreational catch in Florida, but is considered a potential competitor for food and spawning space with native centrarchids or sunfishes (USGS).

Impacts in Australia: Unknown. None reported to date. Known to be present in only two locations in northern Queensland.

(h) Three-spot cichlid (*Cichlasoma trimaculatum*)

Indigenous range: Tropical, equating to water temperatures between 21 and 30°C. Native to Central America, specifically catchments draining into the Pacific from Mexico to El Salvador.

Introduced range: Singapore (FishBase), and Florida from where it was subsequently eradicated (USGS).

Maximum size: Can reach up to 36.5 cm TL, though the maximum length reported for a female of this species 25 cm TL (FishBase).

Habitat preferences: Inhabits lakes and the slow moving waters of the lower river valleys and prefers mud and sand bottoms where it lives among the roots and weeds.

Environmental tolerances: Has relatively broad temperature tolerances. Shafland & Pestrak (1982) reported a lower lethal temperature for this species of 10.9˚C. This could allow the species to occur in many parts of Australia but temperatures for spawning and egg survival may result in a narrower geographic range. Is more likely to thrive in lentic conditions rather than faster flowing streams with rocky substrates.

Behaviour: Is one of the most aggressive cichlids, widely aggressive to conspecifics, and digs a lot at spawning time (Baensch & Riehl 1993).

Reproduction: Little known, but is a clutch guarder and produces c. 1000 eggs; reaches maturity at 80-100 mm length.

Generation time: Minimum population doubling time 1.4 - 4.4 years, which suggests a medium level of resilience (FishBase).

Diet: Has a broad diet including small fish, aquatic macro-invertebrates and both aquatic and terrestrial insects.

Likelihood of natural dispersal: Reaches a relatively large size, so potentially has enough mobility to cope with higher flow velocities, though its preference for slower moving water might suggest otherwise. Only found in fresh water (FishBase), and so may be unlikely to move unassisted between waterways connected by reaches of brackish water. Invasiveness rated as very high by Arthington et al. (1999) and moderate by Bomford & Glover (2004).

Risk of human spread: Is of no interest to commercial fisheries (FishBase) and is unlikely to be introduced to new waterways by coarse fish anglers as a targeted species though, like other established ornamental species, it could be introduced through use as live bait. The volume of fish sold in Australia is relatively low and it is of medium importance to the industry (Table 7.1). The risk of spread by humans is therefore low.

Impacts overseas: FishBase regards this species as harmless.

Impacts in Australia: Unknown. None reported to date. Known to occur in only one location in Australia to date (the Hinze Dam, southeast Queensland).

(i) Jack Dempsey (*Cichlasoma octofasciatum*)

Indigenous range: Tropical, latitudes 14-21°N (corresponding with ambient water temperatures of 22-30°C). Native to North and Central America, specifically on the Atlantic slope from southern Mexico (Papaloapán River) to Honduras (Ulua River).

Introduced range: Thailand, Florida in the USA (FishBase, USGS).

Maximum size: Up to 25.0 cm TL (FishBase).

Habitat preferences: Occurs in lentic systems, ranging from swampy areas with warm, murky water to weedy, mud-bottomed and sand-bottomed canals and drainage ditches, to slow moving waters of lower river valleys and coastal plains (FishBase).

Environmental tolerances: May have relatively narrow salinity tolerances, since it is found only in fresh water. Also has narrow pH tolerances of 7-8 (FishBase), but Shafland & Pestrak (1982) reported a lower lethal temperature for this species of 8.0˚C so it may have a relatively broad temperature range. This could allow it to survive in many parts of Australia, but minimum temperatures for spawning and egg survival may restrict its geographic range. Can also tolerate low oxygen conditions.

Obordo & Chapman (1997) found that it can tolerate hypoxic conditions and that it undertakes air surface respiration when oxygen levels are very low (< 5mm Hg). Its ability for metabolic depression and large gills relative to body size, compared with other fish species, aids the ability to tolerate low oxygen conditions. The capacity to tolerate low oxygen conditions could facilitate establishment in the degraded waterways often found in urban areas. It may, however, be at increased risk of avian predation when undertaking air-surface respiration behaviour (Obordo & Chapman 1997).

Behaviour: Parents incubate eggs and guard young (FishBase), so are likely to undertake aggressive behaviour towards other species or con-specifics during breeding times. Named after a famous American fighter because of its pugilistic temperament (Midgalski & Fichter 1977). A laboratory study conducted by Ratnasabapathi et al. (1992) on the effects of water temperature on aggressive behaviour demonstrated that this behaviour was positively correlated with water temperature, with statistically higher levels of aggression exhibited at 30°C than at 26°C. They believed that this may have been related to the stimulation of convict cichlids to establish territories and spawn at about 30°C.

Reproduction: Between 500 and 800 young per spawning; eggs laid on the substrate, but preferred substrate not specified (FishBase).

Generation time: Minimum population doubling time less than 15 months, which FishBase equates to a high degree of resilience. Time required to reach breeding maturity is less than 1 year and can undertake multiple spawning in a given year (FishBase).

Diet: Reasonably broad. Includes worms, crustaceans, insects and fish (FishBase).

Likelihood of natural dispersal: Brackish water may be a barrier to unassisted dispersal given that it is only found in freshwater systems (though stenohalinity should not be assumed). Obordo & Chapman (1997) suggest that this species' physiological adaptations to cope with low oxygen conditions are possibly traits allowing them to access wetland areas, temporary ponds and other habitats that experience large diel fluctuations in oxygen concentrations. These traits, and the documented behaviour of moving between permanent and ephemeral water bodies, could also allow them to readily extend their range in Australia. Use of ephemeral pools that form after flooding could provide refugia, and this could allow these fish to bypass brackish water when inundation of ephemeral pools by fresh water during floods creates access to new waterways. Invasiveness is rated as very high by Arthington et al. (1999) and high by Bomford & Glover (2004).

Risk of human spread: Related mainly to release into the wild of unwanted fish by aquarists unaware of the risks. The volume of fish sold in Australia is relatively low and it is of medium importance to the industry (Table 7.1). The risk of spread by humans is therefore low.

Impacts overseas: Impacts are not described, though FishBase regards this species as harmless. Riehl and Baensch (1991) describe it as "territorial, intolerant and a biter" in captivity.

Impacts in Australia: Unknown. None reported to date. Known to occur in only one location in Australia to date (Angourie, northern New South Wales).

(j) Red devil (*Amphilophus labiatus*)

Indigenous range: Tropical regions (temperature range 28 – 33°C); native to Central America: Atlantic slope of Nicaragua, in Lakes Nicaragua and Managua. (FishBase).

Introduced range: Established in Hawaii, Singapore (FishBase).

Maximum size: 24.0 cm TL (FishBase).

Habitat preferences: Mostly lentic habitats (i.e., lakes), rarely in slow-flowing rivers (FishBase).

Environmental tolerances: Believed to be restricted to fresh water, though cichlids generally are recognised as having some tolerances of brackish-water salinities; seems unlikely to disperse between sites where migration spanning brackish or marine water is required, though this should not be assumed.

Behaviour: No information found.

Reproduction: Is a nest guarder with 600-700 eggs; little else reported.

Generation time: Minimum population doubling time 1.4 - 4.4 years; recorded as having a medium level of resilience (FishBase).

Diet: Moderately broad given its benthopelagic feeding habitats. Feeds on small fish, snails, insect larvae, worms and other bottom-dwelling organisms (FishBase).

Likelihood of natural dispersal: Inhabits lakes and rarely enters rivers, therefore may be relatively easy to contain where populations have established in Australia. Temperature tolerances suggest mostly northern. Invasiveness rated as high by both Arthington et al. (1999) and Bomford & Glover (2004).

Risk of human spread: No explicit risk beyond the usual concerns about behaviour and attitudes of aquarists who seem not to understand the potential risks. Does not seem highly favoured as an aquarium species, at least internationally. The volume of fish sold in Australia is relatively low and it is of medium importance to the industry (Table 7.1). The risk of spread by humans is therefore low.

Impacts overseas: FishBase regards this species as harmless.

Impacts in Australia: Unknown. None reported to date. Known to be present in only three locations to date (the Ross River in northern Queensland, the Hinze Dam in south-east Queensland, and the Hazelwood Power Station ponds in Victoria).

(k) Midas cichlid (*Amphilophus citrinellus*)

Indigenous range: Tropical, latitudes 8-15°N (corresponding with ambient temperatures of 23-33°C). Native to Central America: specifically the Atlantic slope of Nicaragua and Costa Rica (San Juan River drainage, including Lakes Nicaragua, Managua, Masaya and Apoyo) (FishBase).

Introduced range: Hawaii, Singapore, Florida; possibly Philippines (FishBase).

Maximum size: 24.4 cm TL (FishBase).

Habitat preferences: Found mostly in lakes; uncommon in rivers, and only where slow-flowing. In native range, this species lives in box-cut canals with rocky vertical sides.

Environmental tolerances: Restricted to fresh water (FishBase), so unlikely to disperse between sites where movement through brackish or marine water is required.

Behaviour: Nothing found.

Reproduction: Rocky crevices used for spawning and protection of the young; availability of this habitat may restrict ability to establish new populations. Spawning frequency unrecorded; 300-1000 eggs per spawning event (FishBase).

Generation time: Minimum population doubling time 1.4 - 4.4 years. FishBase regards this as a medium level of resilience.

Diet: Fairly broad, mostly aufwuchs, snails and small fishes but will also consume insect larvae, worms and other bottom-dwelling organisms.

Likelihood of natural dispersal: Will penetrate into rivers where the water is slow-flowing or tranquil (FishBase), so could undertake unassisted dispersal during low flow conditions where it has access to river systems. Temperature tolerances suggest mostly northern potential distribution. Invasiveness rated as very high by Arthington et al. (1999).

Risk of human spread: No explicit risk beyond concerns about thoughtless release. The volume of fish sold in Australia is relatively low and it is of medium importance to the industry (Table 7.1). The risk of spread by humans is therefore low.

Impacts overseas: FishBase regards this species as harmless.

Impacts in Australia: The red devil is included in the proposed grey list for ornamental fish indicating that it is of some concern but that more information is required (DAFF 2005). Currently known to be present in only one location (Ross River, northern Queensland).

(l) Convict cichlid (*Archocentrus nigrofasciatus*)

Indigenous range: Tropical, latitudes 8-15°N (corresponding with ambient temperatures of 20–36°C). Native to Central America, ranging from Guatemala to Costa Rica (Tárcoles River) on the Pacific side, and from Honduras (Aguan River) to Panama (Guarumo River) on the Atlantic side of Central America (FishBase).

Introduced range: Canada (in thermal waters), Hawaii, Japan, United States.

Maximum size: 10.0 cm SL (FishBase).

Habitat preferences: Inhabits flowing water from small creeks and streams to the shallows of large, fast-flowing rivers. Prefers rocky habitats and finds sanctuary in the various cracks and crevices provided by this type of environment, or among roots and debris. Can also occur in warm pools of springs and their effluents (FishBase).

Environmental tolerances: Fairly narrow. Found only in fresh water at pH 7-8. Requires a high water temperature to maintain itself, which may limit its Australian range. A study by Winckler & Fidhiany (1999) demonstrated that the exposure of this species to UVA radiation resulted in metabolic depression, which allowed exposed individuals to tolerate a wider range of temperatures. This may allow it to become established over a wider geographic range in Australia, provided appropriate UVA irradiation conditions prevail. Exhibits anti-predator behaviour cued by chemicals released into the water column when injuries occur to conspecifics, or when conspecifics become frightened (Wisenden & Sargent 1997). Another laboratory study (Alemadi & Wisenden 2002), found that predator-avoidance behaviour of the young, such as schooling or area avoidance occurred despite parental care. Ratnasabapathi et al. (1992) reported that convict cichlids spawn at about 30°C. Therefore this species' reproduction may be limited to sub-tropical and temperate regions of Australia.

Behaviour: Young tend to school as an anti-predator fright response (Alemadi & Wisenden 2002). Both parents defend their territory and protect young during breeding activities (Townshend & Wootton 1984). Keenleyside et al. (1985 *cited in* Beeching 1992) reported that larger males may be more aggressive during the spawning season and therefore present a risk to smaller native fish species compared with larger ones, at this time. However, the cost of defending a territory may outweigh the benefit of having a larger territory for this species (Praw & Grant 1999) and this may limit aggressive behaviour to other species. The extent to which this is manifested in terms of changes in the relative abundance of native species or of community structure may depend on the extent to which the patch size (local habitat occupied by individuals) of convict cichlids is regulated by the combination of food resource availability and the territorial defence effort required. If larger patch sizes are required, or can be easily defended, the scale of impacts of this species on native fish species could increase.

Reproduction: Is described as substrate-spawning by Townshend & Wootton (1984), though more specific information on the types of substrates preferred was not provided. Fecundity according to FishBase is between 100 and 150 offspring per spawning event. McKay (1984) indicated 130-400 based on studies by Sterba (1973) and Cadwallader & Backhouse (1983). Can undertake multiple spawning events within a given year; however, Arthington & Cadwallader (1996) reported 600-3,300 eggs, laid on submerged logs and debris which the adults clean prior to spawning. Eggs are guarded by one or both parents, which also care for the young (Townshend & Wootton 1984). Parents care for eggs and defend nests for up to 4-6 weeks (Alemadi & Wisenden 2002).

Generation time: Minimum population doubling time of less than 15 months. FishBase regards this species as having a high resilience. A study by Townshend & Wootton (1984) found that clutch sizes increased and times between spawning and egg size decreased with reduced food rations independent of changes in weight. Thus, minimum doubling time for wild populations of this species are likely to be affected by food availability at sites where they have become established.

Diet: Feeds on worms, crustaceans, insects, fish and plant matter (FishBase). Trujillo-Jimenez (1998) reported this species as omnivorous, with carnivorous tendencies following a study in the Amacuzac River in Mexico, where animals constituted around 64% of stomach contents. Her study found 26 different prey items among the stomach contents of this species, including dipteran larvae (simuliids and ephemeropterans), though higher plant debris was the most frequently represented item. Although this study sheds little light on dietary preferences, convict cichlids were fed and readily consumed two species of fish (the glowing tetra, *Hemigrammus erythozonus* and swordtail, *Xiphophorus hellerii*) in a predator recognition behaviour study conducted by Brown & Godin (1999).

Likelihood of natural dispersal: Nothing explicit points to a high likelihood of natural dispersal apart from constraints relating to salinities in estuaries and beyond river mouths. Temperature tolerances suggest a mostly northern potential distribution. Invasiveness rated as high by Bomford & Glover (2004).

Risk of human spread: Behaviour of aquarists poses the greatest risk. Englund & Eldredge (2001) suggested a role for aquarists in the liberation of this species in Hawaii. Existing populations are based on such liberations and are indicative of public attitudes to the release of aquarium fish. In Australia, the volume of fish sold is relatively low and it is of medium importance to the industry (Table 7.1). The risk of spread by humans is therefore low.

Impacts overseas: FishBase regards this species as a potential pest. Trujillo-Jimenez (1998) reported correlation-based evidence that *C. nigrofasciatus* displaced a socially-important cichlid that constituted a forage item for local people in Mexico. She concluded that although differences in the feeding behaviour and apparatuses of the two species would tend to reduce dietary overlap, it may nevertheless occur under certain circumstances (e.g., at times when prey species were restricted). This reinforces the need for dietary studies comparing the diet of this and other established ornamental species with Australian native fish species over a range of environments and seasons. Englund & Eldredge (2001) report that in Hawaiian streams "native aquatic species are non-existent or rare where convict cichlids occur", and cited its use as a bait by anglers as responsible for increasing its range.

Impacts in Australia: Unknown. None reported to date. Known to occur in only two locations to date (the Ross River near Townsville, and the Hazelwood Power Station ponds, Victoria).

(m) Blue acara (*Aequidens pulcher*)

Indigenous range: Tropical, latitudes 5-11°N (corresponding with ambient water temperatures of 18-23°C). Native to Central and South America: Trinidad and Venezuela (FishBase).

Introduced range: Indonesia; perhaps Philippines (FishBase).

Maximum size: Up to 16.0 cm TL (Fishbase) but some aquarists report a maximum size of 20 cm. Males grow bigger than females (FishBase).

Habitat preferences: Inhabits turbid standing waters as well as clear, free-flowing streams.

Environmental tolerances: Only found in fresh water, so may have relatively low salinity tolerances (but this should not be assumed); can tolerate highly turbid conditions. Has moderate pH range between 6.5 and 8 (FishBase) so can tolerate moderately acidic as well as basic waters. Has plasticity in the range of water body and hydrological regimes it can inhabit. Although it occurs in latitudes corresponding to a temperature range of 18-23ºC, in aquariums it prefers warmer waters (21-28ºC).

Behaviour: Nothing reported.

Reproduction: Unknown, but perhaps not too specific given that this species reproduces readily in captivity (FishBase).

Generation time: Not described**.**

Diet: Moderately broad; is known to feed on worms, crustaceans and insects (FishBase). Was fed a diet of *Poecilia reticulata* (guppy) during laboratory experiments conducted by Krause & Godin (1995) and Kelley & Magurran (2003). The former conducted experiments to test whether feeding position of prey affected the rate of predation. They found that this species preyed preferentially on guppies that were foraging in a head-down feeding mode than on those foraging in a horizontal feeding mode. Guppies are a natural food item for blue acara (Krause & Godin 1995) and brightly coloured guppies are particularly vulnerable to predation (Godin& McDonough 2003). It may be that similar sized, brightly coloured native fish occupying the same position in the water column and/or trophic niches as guppies may be more at risk than others. Again, such a hypothesis would need to be formally tested before any firmer conclusions can be drawn. Another possibility arising from the above finding is the potential for blue acara to be used as predators to control or eradicate very confined populations of guppies. However, the blue acara is considered a less significant predator of guppies than other co-existing fish species in its native range (Kelley & Magurran 2003). The potential risks and costs of using this control method would need careful consideration. It is reported to root up the bottom making it difficult to keep plants in aquaria (Midgalski & Fichter 1977). Such behaviour in the wild could, if the blue acara was sufficiently abundant, lead to habitat changes affecting the native fauna in some shallow water environments.

Likelihood of natural dispersal: Higher flows may not restrict unassisted dispersal of this species in Australia, given what we know of where it can occur. Is only found in fresh water, so may not move unassisted between water bodies connected by brackish reaches (though low salinity tolerance should not be assumed, given known euryhalinity of some cichlids). Invasiveness rated as very high by Arthington et al. (1999) and moderate by Bomford & Glover (2004).

Risk of human spread: Likely to be related, at least in part, to how widely the species is kept by aquarists. In Australia, the volume of fish sold is relatively low and it is of medium importance to the industry (Table 7.1). Reported to have been used for mosquito control in the USA.

Impacts overseas: FishBase regards this species as harmless.

Impacts in Australia: Unknown. None reported to date. Known to be present in several streams around Brisbane and in the Hazelwood Power Station ponds, Victoria).

(n) Green swordtail (*Xiphophorus hellerii*)

Indigenous range: Tropical to subtropical, at latitudes 12°-26°N (corresponding with ambient water temperatures of 22-28°C). Native to North and Central America, specifically Rio Nantla, Veracruz in Mexico to northwestern Honduras (FishBase).

Introduced range: Established in more than 20 countries in Asia, Africa, Caribbean, Middle East, Indian Ocean Islands. Widely established in the USA (e.g. Hawaii, Texas, Colorado, Florida, California and in some geothermal waters in Idaho).

Maximum size: Males can grow to 14 cm TL, females to 16 cm TL (FishBase, USGS).

Habitat preferences: Found mainly in rapidly flowing streams and rivers, preferring heavily vegetated habitats, but also occurs in warm springs and associated streams, weedy canals and ponds (FishBase).

Environmental tolerances: Has relatively broad salinity tolerances, and can occur in fresh or brackish water. Little known about temperature tolerances. Has broad oxygen tolerances, being able to survive in low dissolved oxygen conditions through surface air breathing (Arthington et al. 1986, *cited in* Morgan & Gill 2001). Appears to tolerate a range of hydrological conditions provided that vegetation cover is available. Its affinity with vegetation cover may also indicate that it is vulnerable to predation in open water, though this remains to be tested. Has relatively narrow pH tolerances, 7-8. Apart from this, the species has some plasticity and can survive in man-made or modified habitats such as urban streams (Arthington et al. 1983).

Behaviour: Males are aggressive towards each other under aquarium conditions (FishBase). Morgan & Gill (2001) cite Franck & Robowski (1993) as reporting that *X. hellerii* form long-term hierarchies and are, to an extent, territorial. Males spend much of their time fighting with conspecifics. Such aggression may or may not extend to native species under field conditions and this remains to be tested. Franck and Robowski (1993, *cited in* Morgan & Gill 2001), believe that such interactions were at least possible and seem likely, given reports that it is capable of dominating the aggressive and highly successful invader, *Gambusia holbrooki* (Arthington et al. 1996, *cited in* Morgan & Gill 2001). Sterba (1966), however, regarded this species as a peaceful, but lively fish in captivity. The aggressive status of *X. hellerii* and the potential for it to have impacts on Australian native fish clearly needs to be evaluated as soon as possible.

Reproduction: Is a livebearer, producing repeated batches of 20-200 young (FishBase). Spawning may be a monthly event and occur all year round provided water temperatures are suitable. Is said to undergo sex changes from female to male (Sterba 1966), though Riehl & Baensch (1991) claim that this is unsubstantiated. If there is sex reversal, there is the potential to form breeding populations even if only females are present initially.

Is non-migratory (FishBase), so does not need to move between waterways to spawn. The reproductive cycle of females ceases when water temperatures fall below 15˚C (Milton & Arthington 1983 *cited in* Morgan & Gill 2001). Such conditions are only met in winter in the subtropical and temperate regions, so it is capable of breeding for extended periods in much of Australia. A testament to this is the fact that 30% of females *X. hellerii* were pregnant in every month of the year except June in the Brisbane region (Milton & Arthington 1983 *cited in* Morgan & Gill 2001).

Generation time: Minimum population doubling time of less than 15 months, which FishBase equates with a high degree of resilience. Time required to reach breeding maturation is less than 1 year (FishBase).

Diet: Relatively broad; feeds on worms, crustaceans, insects and plant matter (FishBase). Arthington (1989) in a study in Queensland found that dipteran larvae were among the dietary components, as were oligochaete worms, molluscs, filamentous algae, diatoms and fish (based on the presence of fish scales), though plant material was preferred. Kailola (2000) considered the species to be omnivorous.

Likelihood of natural dispersal: Allen et al. (2002) attributed establishment to "disposal or intentional release of aquarium pets, or possibly flooding of outdoor ponds." Merrick & Schmida (1984) cite "environmental changes in the Brisbane area" as having advantaged this species. This species' broad salinity tolerance range means that its unassisted range expansion in Australia will probably not be restricted in open systems where freshwater streams are connected by brackish reaches. It is also able to tolerate higher flow conditions compared with other established poeciliids, suggesting that lentic conditions are not likely to hinder its unassisted range expansion provided that streams have reasonable aquatic vegetation cover. However, its non-migratory life mode may limit the rate of unassisted spread in Australia. Invasiveness rated as very high by both Arthington et al. (1999) and Bomford & Glover (2004).

Risk of human spread: Used for genetics research. It has been spread quite widely in Australia, presumably from releases by aquarists, and there seems no reason to believe its spread will not continue. In Australia, the volume of fish sold is relatively high and it is of high importance to the industry (Table 7.1) so the risk of further spread is also high.

Impacts overseas: It has been implicated in the decline of Utah sucker (*Catostos ardens*) in a Wyoming thermal spring (Courtenay et al. 1988) and in the decline of damselflies in Hawaiian waters (Englund 1999).FishBase regards this species as a potential pest and states that several countries report adverse ecological impact after introduction. No further details are provided.

Impacts in Australia: The presence of large numbers of swordtails has been linked with suppression of native fish species in Australia (Arthington 1989). Kailola (2000) reported a negative relationship between swordtail abundance and that of seven native fish species. Furthermore, an examination of this species in the Irwin River found that it was the only fish present suggesting that other species had been displaced (Morgan & Gill 2001). Because of its omnivorous diet (Arthington 1989), its fast breeding capacity, its lack of environmental constraints and especially its ability to coexist with gambusia (Milton & Arthington 1983), Morgan & Gill (2001) concluded that the green swordtail should be declared a pest species. However, the evidence of impacts on native fish in Australia is still sparse and correlative rather than mechanistic. The link between cause and effect needs to be confirmed by obtaining further information on impact mechanisms. This species is rated as a very high-risk species under the Bomford & Glover (2004) risk assessment model. It is also rated as having a very high potential for establishment in Australian waterways under the Arthington et al. (1999) risk assessment model, especially in northern and central latitudes. Currently recorded from eight widely spaced locations throughout Australia including Western Australia, Northern Territory, Queensland, and New South Wales.

(o) Platy (*Xiphophorus maculatus*)

Indigenous range: Tropical and sub-tropical, latitudes 17-23°N (corresponding with ambient water temperatures of 18-25°C). Native to North and Central America, specifically Ciudad Veracruz, Mexico to northern Belize.

Introduced range: Established in at least 12 countries including Africa, Asia, Canada, the Caribbean and the USA (FishBase). In the USA it occurs in Florida, Colorado, Hawaii and Montana (USGS).

Maximum size: Can grow up to 6.0 cm TL (FishBase).

Habitat preferences: Occurs in warm springs, canals and ditches with typically slow-moving water, silt bottoms and weedy banks (FishBase).

Environmental tolerances: Has limited temperature, salinity and hydrological tolerances, and is found only in warm, slow-flowing fresh water. Also seems restricted to waterways with silt bottoms and vegetation cover. Has a relatively restricted pH range of 7-8.

Behaviour: Sociable and non-aggressive in aquaria.

Reproduction: Is a livebearer that reproduces easily some plasticity in spawning requirements; is non-migratory, so does not need to move between waterways to spawn (FishBase). Time required to reach breeding maturation is less than 1 year (can attain sexual maturity after 3-4 months). No information is given in FishBase about spawning frequency. Hybridises readily with the green swordtail (USGS).

Generation time: Minimum population doubling time is less than 15 months, which give the species high resilience (FishBase).

Diet: Relatively broad, feeding on worms, crustaceans, insects and plant matter (FishBase). Arthington (1989) found that crustaceans such as atyid and caridian shrimp were among the foods consumed by these fish in streams in the Brisbane region, as was a range of aquatic and terrestrial insects and algae. Algae were a minor component of the diet of this species, despite being a more major component of the diet of this species in an Indonesian lake (Green et al. 1978, *cited in* Arthington 1989). Such diverse findings probably reflect its trophic plasticity.

Likelihood of natural dispersal: Restricted tolerances for water temperature, salinity and hydrological conditions, coupled with the fact the *X. maculatus* is non-migratory (FishBase), are likely to limit the rate of unassisted range expansion in Australia. This could also aid in the control and eradication of this species. Invasiveness rated as very high by both Arthington et al. (1999) and Bomford & Glover (2004).

Risk of human spread: Depends on popularity of species among aquarists and their attitudes towards release. In Australia, the volume of fish sold is relatively high and it is of high importance to the industry (Table 7.1). The risk of spread is therefore high.

Impacts overseas: FishBase regards this species as a potential pest and states that several countries report adverse ecological impact after introduction. No further details are provided in FishBase. This species has, along with the green swordtail, been implicated in the decline of damselflies in Hawaiian waters (Englund 1999).

Impacts in Australia: Unknown. None reported to date. Now widely distributed in eastern Queensland but only two populations in the Northern territory.

(p) Sailfin molly (*Poecilia latipinna*)

Indigenous range: Subtropical and temperate, latitudes 16-40ºN (corresponding with ambient water temperatures of 20-28 ºC). Native to North America, specifically Cape Fear drainage in North Carolina, United States south to Veracruz in Mexico (FishBase).

Introduced range: Countries of Asia, Africa, Canada, some Caribbean islands, Guam, Hawaii, Middle East, New Zealand (geothermally influenced streams), South America (FishBase) and widely in the USA (e.g., Arizona, California, Colorado, Montana, Nevada and Texas – USGS).

Maximum size: Males of this species can reach 15 cm TL, while females can reportedly grow up to 10 cm (FishBase).

Habitat preferences: Occurs in ponds, lakes, sloughs and quiet, often vegetated backwaters or slow-flowing reaches of streams. Has been found in abundance in artificial habitats such as ditches and tidal canals. Generally found in water bodies less than 1 m deep (FishBase).

Environmental tolerances: Broad salinity tolerance means that it can occur in fresh and brackish water (FishBase). Ability to colonise man-made habitats such as ditches and drains suggests flexibility in terms of habitat requirements, though its restriction to still or slow-flowing waters suggests that spread may be limited by hydrological conditions. Frequent association with vegetation may indicate vulnerability to predators in open water, though this remains to be tested. Reported to require a pH of 7-8.5 in aquaria, so may be restricted mainly to neutral and basic waters.

Behaviour: Sociable and non-aggressive in aquaria.

Reproduction: Is a livebearer that can produce between 10 and 300 offspring per spawning (FishBase), and reproduces repeatedly. Is non-migratory (FishBase), so does not need to move between waterways to spawn.

Generation time: Minimum population doubling time of less than 15 months gives this species a high resilience (FishBase). Time required to reach breeding maturation is less than 1 year and can undertake multiple spawnings in a given year (FishBase).

Diet: Relatively restricted. Is considered benthopelagic, and feeds mainly on algae (FishBase). Courtenay & Meffe (1989) indicated that it is mainly herbivorous.

Likelihood of natural dispersal: This species' broad salinity tolerance means that its unassisted range expansion in Australia will probably not be restricted in open systems where freshwater streams are connected by brackish reaches. However, its non-migratory life mode may limit its rate of unassisted spread in Australia. Furthermore, it is found only in slow-flowing or still waters, suggesting that higher stream flows may hinder unassisted range expansion in Australia. These factors may aid in the ability to control or eradicate populations of this species. Invasiveness rated as very high by both Arthington et al. (1999) and Bomford & Glover (2004).

Risk of human spread: A distinctive feature of this species is that rearing in natural habitats produces much larger 'sail-fins' in the males, and this is an incentive to try and establish feral populations so that they can be harvested at a later data for sale. In Australia, the volume of fish sold is relatively high and it is of high importance to the industry (Table 7.1). The higher value placed on fish with large 'sail-fins', coupled with the high demand for this species, increases the risk that it will be relased into the wild.

Impacts Overseas: FishBase regards this species as a potential pest and states that adverse ecological impacts have been reported after its introduction, but other information on the impact mechanisms, consequences or manifestations is not provided. It is reputed to be responsible for the decline of the desert pupfish (*Cyprinodon macularius*) in California (US Fish & Wildlife Service 1983). Along with other introduced poeciliids it has also been blamed for the decline of damselflies in Hawaii (Englund 1999).

Impacts in Australia: Regarded as having a potential for adverse effects on endemic species of fish (Arthington 1989). While its breeding and habitat plasticity may seem to point to this species having reasonable invasive capacity, the relatively restricted diet and absence of records of aggressiveness appear to make competition for food with native fish species, competition for space by means of aggressive territoriality, or predation, unlikely impact mechanisms. Given that algae are a major component of its diet, it may have adverse impacts on native fish species through habitat modification (e.g., through degradation of their habitat or that of their preferred prey). Recorded from three geogrpahically spaced locations in eastern Queensland

(q) Guppy (*Poecilia reticulata*)

Indigenous range: Tropical, latitudes 2-14°N (corresponding with ambient water temperatures of 18-28°C). Native to South America, specifically Venezuela, Barbados, Trinidad, northern Brazil and the Guyanas (FishBase).

Introduced range: Very widespread in > 40 countries, from Russia to New Zealand (in geothermal waters) and the Americas, Asia, Africa and Europe.

Maximum size: Males grow to 3.5 cm SL, females to 5.0 cm SL (FishBase) but reports of 6 cm in aquaria.

Habitat preferences: Occurs in warm springs and associated streams, weedy ditches and canals. Found in various habitats, ranging from highly turbid ponds, canals and ditches at low elevations to pristine mountain streams at high elevations (FishBase).

Environmental tolerances: Has wide salinity tolerances, but requires fairly warm temperatures (23-24 °C) and quiet vegetated water for survival (FishBase). However, it has been found in many temperate countries (Welcomme 1988), so its actual temperature tolerances could be much greater than FishBase suggests. *P. reticulata* also appears to tolerate a range of turbidities ranging from clean water in pristine streams to highly turbid canals and ditches. Also seems to have plasticity in terms of its habitat requirements given the range of water bodies it has been found in, though the frequent association with vegetation suggests that this species may be vulnerable to predation in open water. Has relatively narrow pH tolerances of 7-8 (FishBase). Colourful males are vulnerable to predation, but can overcome this by changing colour to become less colourful when predators are prevalent (Gomez & Ferriz 2002). This species also has the capacity to increase its growth rates when under high predation pressure, however, this response to predation will depend largely on food availability (Arendt & Reznick 2005).

Behaviour: May form schools as part of its predator avoidance behaviour (Reznick & Endler 1982). In areas where it has been introduced overseas it has had a negligible effect on native fish populations (FishBase). This probably indicates a lack of aggressiveness, but it is not possible to rule out aggressiveness of this species towards Australian native fish without further investigation.

Reproduction: Is a livebearer, typically producing batches of between 20 and 40 offspring, but sometimes many more. Undergoes sex change from female to male

(Sterba 1966) when there are few or no males in populations, and so has the potential to form breeding populations even if a population is originally all female. Is non-migratory (FishBase), so does not need to move between waterways to spawn. Male colouration provides a visual cue for female choice (Gomez & Ferriz 2002).

Generation time: Minimum population doubling time less than 15 months, which FishBase equates with a high degree of resilience. Time required to reach breeding maturation is between 0.16 and 0.25 years (males may mature at 2 months and females at 3 months of age) and this species can spawn multiple times per year (FishBase). Reznick & Endler (1982) found that, when in the presence of predators that have this species as a major component of their diet, the young reach maturity more quickly and produce more offspring. Hence, figures given for time to reach maturity and clutch size given in FishBase may be underestimates for some sites. They also state that these changes can take place rapidly (within 2.5 years). These findings demonstrate this species' capacity to compensate for natural population control by predators, which would seem to reduce the usefulness of predator introduction as a means of control or eradication of this species.

Diet: Reasonably broad; feeds on zooplankton, small insects and detritus. Has been introduced in some countries for mosquito control, and so is thought capable of feeding on these and other dipteran larvae, but this is unverified. Arthington (1989) found that populations around Brisbane included ants along with chironomid midge larvae (diptera) as major components of their diet. This contrasts with the diet of this species in Trinidadian streams, which comprised larger proportions of aquatic insect larvae, unicellular algae, diatoms and plants (Dussault `& Krumer 1981, *cited in* Arthington 1989). Such diverse findings probably reflect the trophic plasticity of this and other poeciliids (Arthington 1989). Also eats the eggs of other fish (Welcomme 1988), which may be a potential impact mechanism in Australian streams and ponds. This requires further examination.

Likelihood of natural dispersal: This species' broad salinity tolerances mean that its unassisted range expansion is likely in open systems where freshwater streams are connected by brackish reaches. However, its non-migratory life mode may limit the rate of unassisted spread in Australia. Furthermore, it is found mainly in lentic systems, suggesting that higher flow conditions may be a barrier to its unassisted range expansion in Australia. These factors may assist in the control or eradication of populations. Gomez & Ferriz (2002) found that larger, long-tailed individuals of this species tolerate velocities of 15.4 cm s^{-1} for almost an hour at 24°C before fatigue sets in. This is probably more of an adaptation for predator avoidance than for migration (Gomez & Ferriz 2002). *P. reticulata* is known to coexist with predators and to be a favoured food source of others (Reznick & Endler 1982). Little published information exists regarding predation of this species by Australian native fish, so it is difficult to gauge whether its populations might be kept under control by predation. A host of authors (*cited in* Reznick & Endler 1982) have reported that *P. reticulata* is known to become brighter, larger, tends to school less often, and reacts at shorter distances to the presence of predators where different generations are exposed to higher predation pressure. A later study, published by Kelley & Magurran (2003), reached similar conclusions and stated that this adaptive behaviour was a heritable trait. Assessment of morphologies and behaviour at different sites in Australian waterways may, therefore, go some way towards determining the degree to which populations of this species are controlled by predation at the local scale. Populations in geothermally influenced streams near Rotorua in the North Island of New Zealand have not become established in the somewhat colder, ambient waters further downstream (D. Rowe, pers. comm.). Invasiveness in Australia rated as very high by Arthington et al. (1999) and extreme by Bomford & Glover (2004).

Risk of human spread: In Australia, the volume of fish sold is relatively high and it is of high importance to the industry (Table 7.1). The risk of spread by humans is therefore high.

Impacts overseas: FishBase regards this species as a potential pest and states that several countries report adverse ecological impact after introduction, but provided no details.

Impacts in Australia: Unknown. None reported to date. Now spread widely through the eastern waters of Queensland and northern New South Wales. Isolated populations also recorded in western Australia and the Northern Territory.

(r) Caudo (*Phalloceros caudimaculatus*)

Indigenous range: Tropical latitudes corresponding with ambient water temperatures of 20-24°C. Native to South America, specifically Rio de Janeiro, Brazil southward to Uruguay and Paraguay (FishBase).

Introduced range: Malawi, New Zealand (establishment uncertain).

Australian range: Low elevation waters of coastal New South Wales (Sydney, Newcastle); also near Perth in Western Australia.

Maximum size: Males reach 3.5 cm SL, while females can grow to 6 cm TL (FishBase; Sterba 1966).

Habitat preferences: Still waters and small ponds and margins.

Environmental tolerances: Relatively broad salinity tolerances, being found in both fresh water and brackish water (FishBase). Sterba (1966) states that this species is resistant to low temperatures and can be kept in unheated aquaria at temperatures of 12-18ºC, but prefers 20-24ºC. Has narrow pH tolerance of 7-8 (FishBase) indicating that it prefers basic to acidic waters.

Behaviour: Is a very peaceful and easily satisfied species (Sterba 1966).

Reproduction: Is a livebearer, with large females producing clutches of up to 80 young per spawning (Sterba 1966). Spawning requirements are minimal. Does not undertake any migration for spawning (FishBase).

Generation time: Minimum population doubling time is less than 15 months, giving this fish a high resilience (FishBase).

Diet: Regarded as omnivorous (FishBase). No further details were given in FishBase other than that it has been introduced for mosquito control. Thus, its diet is likely to include mosquito larvae and pupae and possibly other dipterans.

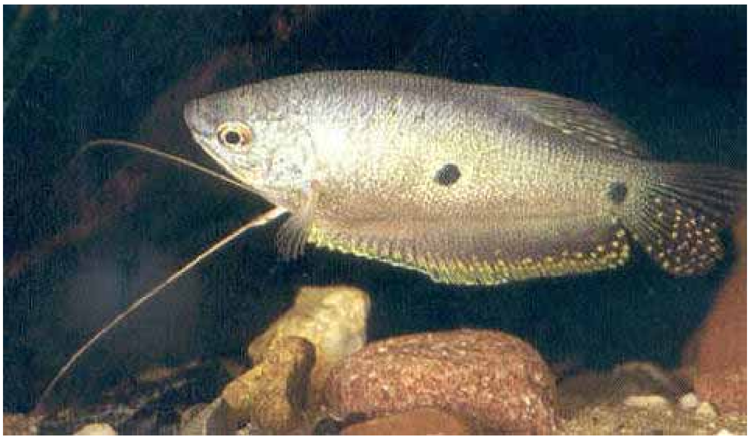
Likelihood of natural dispersal: This species' broad salinity tolerances mean that unassisted range expansion in Australia is likely. Will probably not be prevented from invading freshwater streams connected only by brackish reaches. However, the fact that this species is non-migratory (FishBase), coupled with limited mobility owing to small maximum size might suggest that the rate of unassisted range expansion rate would be slow. This supposition remains to be tested. Invasiveness rated as very high by Arthington et al. (1999) and high by Bomford & Glover (2004).

Risk of human spread: If feral populations are easily established its spread may be rapid, but there is little field evidence for this yet. In Australia, it is of low importance to the industry (Table 7.1) so the risk of spread by humans is low.

Impacts overseas: FishBase regards this species as a potential pest and stated that at least one country reports adverse ecological impact after introduction. No further details are provided, so the impact mechanisms in those countries are unclear.

Impacts in Australia: Observations of this species in the Perth region indicated that it may have displaced *Gambusia holbrooki* from one location (ASFB 2003c). Very similar observations were made by Rowley et al. (2005) for a population in Sydney. If this species has in fact displaced *Gambusia*, which is a well known pest fish species that often displaces small native fish species, then the caudo may well be able to displace small native fish species as well. There is no quantitative evidence for this at present so data on its interactions with other fish are urgently required. It is rated as a high-risk species under the Bomford & Glover (2004) risk assessment model. It is also rated as having a very high potential for establishment in Australian waterways under the Arthington et al. (1999) risk assessment model. Given this high 'invasive' potential plus the probability of impact on small native fish species, concerns over its potential impact are warranted. Currently present in a number of waters around Perth and in two locations in New South Wales.

(s) Three-spot gourami (Trichogaster trichopterus)



Indigenous range: Tropical (latitudinal range 26-10°N), and found at temperatures of 22-28°C; native to southeast Asia: Mekong basin in Laos, Yunnan, Thailand, Cambodia and Viet Nam.

Introduced range: Introduced quite widely elsewhere including Colombia, Dominica, Namibia, Papua New Guinea, Philippines, Sri Lanka (FishBase) and Florida (USGS).

Maximum size: 15.0 cm SL (FishBase).

Habitat preferences: Found in marshes, swamps and canals. Prefers shallow sluggish or standing-water with abundant aquatic vegetation; pH 6-8 (FishBase) indicating tolerance of mildly acidic as well as basic waters.

Environmental tolerances: Restricted to fresh water, so can't disperse between sites where migration through brackish or marine water is required. Relatively narrow pH tolerance range, 6-8 (FishBase). Has auxiliary breathing mechanisms so can tolerate low oxygen conditions (Arthington and Marshall 1999). These authors considered that it has a moderate to high probability of establishment in Australia.

Behaviour: Is benthopelagic and solitary. Regarded as 'peaceful and contented' Sterba (1966) and Riehl &Baensch (1991) in aquaria.

Reproduction: Constructs a bubble nest near the water surface where it deposits 1500-2000 eggs; these are guarded by the male, and hatch in 1.0-1.5 days; larvae are tiny and not robust. Reaches maturity at 70-80 mm length.

Generation time: Minimum population doubling time less than 15 months; multiple spawning per year. This equates with a high degree of population resilience (FishBase).

Diet: Moderately broad, primarily invertebrates including zooplankton, crustaceans and insect larvae; Riehl and Baensch (1991) describe it as "omnivorous".

Likelihood of natural dispersal: In the Mekong, this species has a tendency to undertake migrations from permanent waters bodies to flooded areas and then return to permanent water bodies at the onset of the dry season (FishBase). This behaviour might result in unassisted movements to new locations in Australia should it occur in regions where inundation of floodplains allows its escape from permanent water bodies. This might pose a particular risk of invasion of billabongs. Is regarded as freshwater-limited, which means that dispersal around coasts or in harbours and inlets, in brackish water, is unlikely. Invasiveness rated as very high by Arthington et al. (1999) and extreme by Bomford & Glover (2004).

Risk of human spread: In Australia, this species is of high importance to the industry but volumes sold are not high (Table 7.1). The risk of spread is therefore moderate.

Impacts overseas: FishBase regards this species as harmless, though the basis for this judgement is unclear.

Impacts in Australia: None reported. Populations known to occur in the Ross and Burdekin Rivers in Queensland.

(t) Oriental weatherloach (*Misgurnus anguillicaudatus*)

Indigenous range: Subtropical to cold, temperate latitudes 27-53°N (corresponding with ambient water temperatures of 10-25°C). McMahon & Burggren (1987 *cited in* Lintermans et al. 1990) described this species as eurythermal and suggested that it can thrive in water temperatures from 2-30ºC. Native to Asia, specifically Myanmar and northeastern Asia and southward to central China (FishBase). Greatest densities in southeast Asia and the Malay Archipelago (Sterba 1962).

Introduced range: Eastern Europe, Hawaii, Mexico, Palau, Philippines (FishBase). Established in Oregon and Washington as well as in a number of southern States in the USA (USGS).

Maximum size: Up to 24.8 cm TL (FishBase).

Habitat preferences: Occurs in rivers, lakes and ponds, swamps and rice fields at depths down to 5 m. Prefers cool, slow-flowing waters with muddy or weed covered bottoms. It creates shallow burrows in the substrate where it can hide with only its head sticking out. This behaviour is also associated with ‘overwintering’ (Burchmore et al. 1989), but is also believed to be a form of predator avoidance (Sterba 1962). In Hawaii, where it has been introduced, it has been found living in association with algal/macrophyte mats (FishBase). Tabor et al. (2001) found that it was associated mainly with macrophytes in shallow waters.

Environmental tolerances: Has broad habitat tolerances and is found in a wide range of habitats. Typically regarded as a cool water species (Welcomme 1988). Can tolerate highly turbid and deoxygenated conditions (Burchmore et al. 1990; Welcomme 1988), which allows it to colonise degraded aquatic habitats often found in association with large urban areas. Can also tolerate low oxygen conditions, partly through its ability to swallow air and pass it through its highly vascularised intestine (McMahon & Burggren 1987 *cited in* Lintermans et al. 1990). Can survive in highly modified waters, this being an indication of its habitat plasticity. Ip et al. (2004) noted that it is drought resistant and buries in the mud to avoid dehydration. Lintermans et al. (1990) report that, at one site in the ACT, weatherloach was abundant in association with an artificial soil structure consisting of wire cages containing rock fill. It is even said to be able to tolerate pesticide contamination (Lee & Lee 1985, *cited in* Lintermans et al. 1990), which begs the question how effective chemical control mechanisms will be in eradication or reducing populations. Perhaps more disturbingly, from an invasiveness perspective, is its reported ability to crawl out of water, survive in damp soil and even move across land (comments attributed to A.K. Morrison *cited in* Burchmore et al. 1990).

Behaviour: Not known as aggressive; primarily solitary.

Reproduction: Up to 2000 offspring per spawning (FishBase). Under laboratory conditions has been induced to spawn every 20 days or so for 13 months (Suzuki 1983 *cited in* Burchmore et al. 1990). Whether or not this is replicated in the wild in Australia remains to be determined. Spawning is linked to water levels and access to floodplains in its native range (Tanaka 1999), which might also hold for this species in Australia.

Generation time: Minimum population doubling time is 1.4 - 4.4 years, which FishBase equates with a medium level of resilience.

Diet: Broad. Feeds on worms, small crustaceans, insects, insect larvae, and other small aquatic organisms (FishBase). A study by Katano et al. (2004) found that *M. anguillicaudatus*, along with several other species studied, consumed mainly dipterans (chironomid and ephemeropteran larvae). Tabor et al. (2001) found that it fed mainly on benthic prey such as chironomids and detritus and used chemical stimulation to locate its prey. In Australia, the diet of this species was more or less consistent with that listed in FishBase, but also included detritus as a major component (mean of 36% by volume) (Burchmore et al. 1990). However, this was based on the stomach contents of only 5 individuals, and the list of prey items consumed was probably greater than this, so the relative importance of detritus in the diet may have been exaggerated.

Some workers believe that the preference of this species for muddy substrates is linked to its preference for detritus as a dietary component (Watanabe & Hidaka 1983 *cited in* Lintermans et al. 1990). The relative importance of the various prey items this species consumes in Australia at various times of the year (or at various locations) awaits a more thorough assessment.

Likelihood of natural dispersal: Is probably restricted to fresh water so that brackish water would likely prove an obstacle to the unassisted range expansion of this species. In its native range, it moves via creeks and drainage networks from permanent water bodies to rice fields on the flood plains where it reproduces. It then returns to permanent water bodies to 'overwinter' (Tanaka 1999). Thus, inundation of floodplains in Australia may provide this species with a spawning cue and be a means of dispersal. However, the ability of this species to move across land means that it may spread more readily from one water body to another in low-lying or relatively flat areas. Burrowing activities during overwintering may affect some environments and mean that sampling needs to be carried out in summer, especially if netting or trapping are the main techniques used (Burchmore et al. 1990). Invasiveness rated as very high by Arthington et al. (1999) and high by Bomford & Glover (2004).

Risk of human spread: Is valued as a fishery and aquaculture species in other countries, so has potential for dispersal to new waterways by coarse fish anglers and aquaculturists. The USGS fact sheet on this species indicates that it has been spread within the USA from aquarium facilities, by asian immigrants as a food source and by anglers as a bait fish. Lintermans et al. (1990) stated that it is 'not desired' by Australian anglers. Is also considered for commercial bait harvest in some countries (FishBase) and could be transferred to new waterways in Australia through use as live bait. Allen et al. (2002) attribute establishment to "thoughtless disposal of unwanted fish by aquarists" and escapes when used by anglers as live bait. This species' ability to survive in turbid and deoxygenated conditions may allow easy expansion of range into degraded urban waterways.

Impacts overseas: FishBase regards this species as a potential pest and states that adverse impacts have been reported in at least one country where established. Burchmore et al. (1990) reported assessments from different countries, ranging from 'benign' in mainland USA, to 'uncertain' in Mexico, to 'intermediate' in Hawaii.

Impacts in Australia: Was declared a banned import in 1986 owing to its 'feral habits' (Burchmore et al. 1990), but it is unclear whether this was based on its potential invasiveness alone or incorporated perceived potential environmental impacts. Burchmore et al. (1990) argued that it should be declared a noxious species, but this appears to have been based purely on its successful establishment and the impossibility of eradication, rather than actual adverse ecological impacts. *M.*

anguillicaudatus is designated as a noxious species in the ACT (Lintermans et al. 1990).

Suggested potential impacts relate to this species' habitat preferences, its burrowing activities, diet, competition with native fish for spawning sites, disturbance or predation of eggs, competition with native species for planktonic food (particularly with larval fish), competition for shelter, and habitat alteration (Lintermans et al. 1990b). Lintermans et al. considered these impacts speculative and we have not been able to uncover studies investigating any of these perceived potential impacts.

Evidence of impacts is based primarily on correlative observations, as is almost always true of evaluations of the impacts of invasive species. Lintermans (1993) found that *Galaxias olidus* and oriental weatherloach were never found together in a small stream near Canberra, though the galaxiid was present 150 m upstream. He observed that it was not possible to determine whether this finding reflected competitive exclusion, or exploitation of habitats by weatherloach that were unsuitable for the galaxiid. Arthington & Blühdorn (1995) cited personal comments from Lintermans that there was 'preliminary evidence' of the impacts of weatherloach on the mountain galaxias. Environment ACT (2002) indicated that it may be responsible fore a localised decline of Mountain Galaxias and Western Carp gudgeon in part of the Ginninderra Creek catchment.

This species is rated as a high-risk species under the Bomford & Glover (2004) risk assessment model and is on the proposed noxious species list proposed in the draft strategic approach to the management of ornamental fish in Australia (DAFF 2005). It is also regarded as having a moderate to high probability of establishment under the Arthington et al. (1999) risk assessment model. Now widely distributed in south-eastern New South Wales and Victoria.

(u) Goldfish (*Carassius auratus*)

Indigenous range: Subtropical to temperate latitudes of 22-53°N (corresponding with ambient water temperatures of 0-41°C). Native to Asia, specifically central Asia and China, and Japan (FishBase).

Introduced range: Very widespread from the cold temperate to sub-tropics, all over the world.

Maximum size: Reportedly grows to 59.0 cm TL and 3,000 g in weight. Can live as long as 30 years (FishBase).

Habitat preferences: Inhabits rivers, lakes, ponds and ditches with stagnant or slow-flowing water (FishBase). Tropical populations of this species are generally found at altitudes of 200-1000 m (Welcomme 1988).

Environmental tolerances: Occurs in habitats where there is a wide range of temperatures. Also has broad salinity tolerances, and can tolerate salinities as high as 17 ppt, but can't cope with prolonged exposure above 15 ppt. Occurs in a variety of habitats and tolerates low oxygen conditions associated with stagnant water. Broad tolerates of acidity indicated by pH range of 6 to 8 (FishBase).

Behaviour: Peaceable and sociable showing little aggression.

Reproduction: Lays a few to many thousands of eggs on submerged vegetation; larvae are pelagic; no parental care (FishBase). According to Holcik (1980), this species can breed independently of water levels, and this was one of the factors that contributed to its success in the Danube system compared with native species of carp that need water levels to be high enough to allow access to floodplains to breed. Establishes self-sustaining populations easily in small ponds to large lakes, especially where there is plentiful aquatic vegetation.

Generation time: Minimum population doubling time 1.4 - 4.4 years, which FishBase equates with a medium level of resilience. Maximum time taken to reach breeding maturity is around 1 year.

Diet: Consumes a wide range of foods including plants, small crustaceans, insects, and detritus (FishBase; Arthington and Mackenzie 1997). Goldfish will suck up a mouthful of sediment when feeding on benthos and the sediment is then spat out raising turbidity levels (Richardson et al. 1995). Has also been known to prey on salamander eggs (Monello & Wright 2001), and so may pose a threat to some Australian frog species, though this would require further investigation.

Likelihood of natural dispersal: Is probably unable to disperse between river catchments through coastal, brackish estuaries and seas, but is likely easily to spread into billabongs that are intermittently connected to mainstem rivers where it is present. This is a likely explanation for the species' present broad range in southeastern Australia. Invasiveness rated as very high by Arthington et al. (1999) and extreme by Bomford & Glover (2004).

Risk of human spread: Holcik (1980) demonstrated population explosions of goldfish in the Danube River system corresponding to decreased catches of local predators caused by a combination of fishing pressure, barriers to fish passage, altered hydrological regimes and atypically low water levels that restrict access to breeding sites on the floodplains. Many of the progeny of recently released goldfish often have bright colouration making it vulnerable to predation (A. Moore pers. obs.), but this is likely to result in strong selection against such colouration persisting in wild populations. If fishing pressure reduces predation on goldfish in Australian waterways through removal of piscivores, or if human activities or natural factors affect the access of those predator species to their spawning sites, there is potential for similar population explosions of goldfish in this country. Arthington & Blühdorn (1995) reported that this species was probably being sold as rock lobster bait which, if true, is a cause for concern as conflicts of interest may drive those who profit from this to introduce goldfish into new waterways to guarantee a continued supply. There are anecdotal reports of this species being used as live bait by anglers (Arthington & Blühdorn 1995) which, regardless of legislation introduced in some states to stop this activity, poses another mechanism for further spread in Australia. In Australia, the volume of fish sold is very high (Table 7.1) so the risk of further spread by humans is high.

Impacts overseas: FishBase regards this species as a potential pest and states that several countries report adverse ecological impact after introduction; no further details are provided on the mechanisms or manifestations of these impacts. Welcomme (1988) suggested that the environmental effects of *C. auratus* are somewhat neutral and that its nuisance value has more to do with its capacity to produce stunted populations that are of little use in terms of forage. Crivelli (1995) provides a Mediterranean example of this, whereby populations introduced into Lake Kastoria in Greece resulted in neither increased yields nor income for local fishers, despite representing 80% of their catch.

Other workers have reported increases in turbidity levels and damaged or uprooted aquatic plants in muddy sediment ponds where this species has been introduced, and that their diets have overlapped with some Canadian frog species (Richardson & Whoriskey 1993; Richardson et al. 1995). These workers also stated that avian predation pressure on this species may actually be reduced owing to the role of goldfish in increasing turbidity. Richardson et al. (1995) clearly demonstrated that *C. auratus* was able to increase turbidity levels in muddy pools by 10 times compared with experimental control muddy pools. Goldfish were, however, not able to achieve significant changes in turbidity where experimental pools had sand/gravel beds. The increase in turbidity generated by goldfish in these experimental trials resulted in changes in temperature regimes as well, but did not significantly affect the growth of aquatic plants. Plants were, however, commonly up-rooted by goldfish in this trial, though they were not observed being eaten by these fish. An earlier study by Richardson et al. (1992) demonstrated that turbidity generation by *C. auratus* was positively correlated with body size, so populations of larger-sized fish of this species have the most potential to increase turbidity levels (and by association, have impacts on native fish or their prey).

Archibald (1975) used manipulative laboratory experiments to determine the influence of increasing vertebrate predation pressure on zooplankton communities, using *C. auratus* as the vertebrate predator of choice. The response of zooplankton to different levels of predation pressure in these experiments was complex, with some species increasing in density at low-to-intermediate predation levels, while a decline in most zooplankton species occurred under high predation pressure. This study demonstrates that 1) juvenile *C. auratus* are capable of feeding on elements of zooplankton communities in Australian waterways and, 2) that complex changes in the community structure could occur in conjunction with fluctuating densities of this species. Given that *C. auratus* has been known to numerically dominate some fish communities in Australia, there is potential for zooplankton densities to decline markedly as a result of high predation pressure from this species. This could have cascading ecological effects that impact on Australian native fish, particularly those zooplankton grazers that feed largely in the middle of the water column.

Impacts in Australia: Wager & Jackson (1993 *cited in* Arthington & Blühdorn 1995) and also *in* Arthington & Mackenzie (1997), suspected that *C. auratus* may have adverse impacts on the 'endangered' indigenous trout cod *Maccullochella macquariensis*. A control programme is being undertaken to remove a newly established population in the Vasse River, Western Australia, because of concerns that it will feed on fish eggs, reduce macrophytes, increase turbidity and stimulate blue-green algal blooms (ASFB 2005). Although there is as yet no compelling evidence for such impacts, the concerns are valid, especially in environments where goldfish will proliferate. Pritchard et al. (2004) found that goldfish were a minor component of the fish fauna in the waterways of the Lake Eyre Basin in both 1986-1992 and 2000-2003. Goldfish are regarded as a prescribed non-indigenous fish in Queensland under the Fisheries Act, but this only means that it is illegal for it to be released; it may be kept in aquaria (Arthington & Blühdorn 1995). Is rated as an extremely high-risk species in terms of its invasive potential under the Bomford & Glover (2004) risk assessment model. Also rated as having a very high risk of establishment under the risk assessment model of Arthington et al. (1999). Its existing, broad range may be close to that which it can achieve in eastern Australia, though there is no doubt some potential for the species to occupy many more water bodies within that range. Its current distribution probably reflects the species high popularity and its long presence in Australia (since the 1860s – Allen et al. 2002). Very widely distributed throughout the southeast of Australia, including parts of Tasmania. Also present in lowland waters in the southwest of Western Australia.

(v) Rosy barb (*Puntius conchonius*)

Indigenous range: Subtropical to temperate, latitudes 8-40°N (corresponding with ambient water temperatures of 18-22°C). Native to Asia, specifically Afghanistan, Pakistan, India, Nepal, and Bangladesh (FishBase).

Introduced range: Canada, Colombia, Mexico, Puerto Rico, Singapore (FishBase) and Florida (USGS).

Maximum size: Up to 14.0 cm TL (FishBase).

Habitat preferences: Inhabits lakes and fast-flowing hill streams (FishBase).

Environmental tolerances: Considered to be one of the hardiest of the barbs (FishBase), with broad environmental tolerances. Its fairly restricted latitudinal temperature range may limit its unassisted range expansion in Australia to the subtropics. Has limited salinity tolerance and is only found in fresh water. Tolerates pH between 6 and 8. According to Sterba (1966), water quality conditions are generally unimportant to members of the genus *Puntius*, but these fish prefer water that has been "well matured by plants". The young of this species are vulnerable to damage by copepods (such as *Cyclops* spp.) and ostracods (Sterba 1966), which are likely to occur in most water bodies in Australia. There may, therefore, be potential for producing populations of these invertebrates for use in controlling recruitment of the rosy barb, but this remains to be tested and any side-affects examined.

Behaviour: According to Sterba (1966), members of the genus *Puntius* often occur in large shoals. Such behaviour might also be exhibited by *P. conchonius,* though this requires further investigation. Aggressiveness is probably limited given that this species can be kept together with other small fishes (FishBase) and is considered a peaceful fish (Sterba 1966; Riehl & Baensch 1991).

Reproduction: All members of the genus *Puntius* are oviparous. Young hatch 24-36 hours after eggs are laid and avoid predation by staying in close proximity to the substrate and vegetation for 1-2 days. Parents are notorious spawn-robbers and can devour their young (Sterba 1966). For breeding in aquaria, Sterba (1966) recommends a soft and not too pale substrate with a loose and not too dense screen of floating plants. While this recommendation was not specific to *P. conchonius*, or wild populations of this species, it may be that *P. conchonius* requires these habitat conditions in the field for sustaining its populations. Sterba (1966) states that all fine-leafed plants are suitable for egg deposition. However, breeding of members of the genus *Puntius* is said to be a relatively simple process (Sterba 1966), which might indicate plasticity in their spawning requirements. For some species of *Puntius*, spawning may be triggered by influxes of fresh water in combination with light incidence during mornings (Sterba 1966).

Generation time: Minimum population doubling time less than 15 months and can reach breeding maturation in less than 1 year (FishBase). Sterba (1966) states that aquarium-reared populations of the genus *Puntius* can begin spawning between 9 and 12 months. Considered to have a high level of resilience (FishBase).

Diet: Relatively broad. Food consists of everything from worms, crustaceans and insects to plant matter. Considered to be a gluttonous feeder (Sterba 1966), which may allow it to quickly deplete food resources that might be shared with other species.

Likelihood of natural dispersal: Can tolerate fast-flowing conditions and so may be able to move between waterways. However, it cannot tolerate brackish water, so its unassisted range expansion in situations where water bodies are separated by brackish reaches is unlikely. Invasiveness rated as high by Arthington et al. (1999) and very high by Bomford & Glover (2004).

Risk of human spread: Again, this risk relates to how widely the species is kept in captivity and the behaviour of those who keep them (or no longer wish to keep them!). Even though it is of high importance to the aquarium industry in Australia, the volume of fish sold is moderate (Table 7.1). The risk of spread is therefore moderate.

Impacts overseas: FishBase regards this species as harmless.

Impacts in Australia: Unknown. None reported to date. Currently only reported from streams in and around Brisbane.

(w) White cloud mountain minnow (*Tanichthys albonubes*)

Indigenous range: Tropical, latitudes 9-30°N (corresponding with ambient water temperatures of 18-22°C). Native to Asia, specifically China and Vietnam (FishBase).

Introduced range: Colombia, Madagascar, possibly Canada, Philippines, United States.

Maximum size: Up to 4.0 cm TL (FishBase).

Habitat preferences: Prefers weedy streams and ponds.

Environmental tolerances: Found only in freshwater ecosystems and, being a cyprinid, probably has narrow salinity tolerances. Despite the relatively restricted water temperature ranges in its native range, this species can tolerate water temperatures as low as 5°C (FishBase); also tolerates pH of 6-8, but occurs only in fresh water (FishBase). No information regarding its tolerance to a range of hydrological conditions is provided in FishBase. Seems to need cool winter temperatures.

Behaviour: Peaceable and sociable in captivity (Riehl and Baensch 1991), and likely to be the same in nature.

Reproduction: Likely to have low fecundity owing to small adult size, less then 250 small eggs, but may repeat spawn often; ova spread on substrate.

Generation time: Minimum population doubling time less than 15 months and can reach breeding maturation in less than 1 year (FishBase). Considered to have a high level of resilience (FishBase).

Diet: Relatively restricted. Given its demersal habitat (FishBase) it is likely to prey mainly on benthic species. However, FishBase states that it feeds on zooplankton as well as detritus.

Likelihood of natural dispersal: Can tolerate a range of water temperatures and so has the potential to survive in many parts of Australia (though it could be that winter minimum temperatures in some areas affect its reproductive output or production rates). This is yet to be tested. *T. albonubes* is only found in fresh water, so brackish water may provide a barrier to its unassisted dispersal in open systems where freshwater reaches are connected by brackish water reaches. Invasiveness rated as moderate to high by Arthington et al. (1999) and high by Bomford & Glover (2004).

Risk of human spread: The ease with which this species may establish populations in small ponds may add to the risk of spread. Otherwise, this depends, again, on how often it is kept and the attitudes of aquarists. Its seemingly small size may promote the thought that it is harmless. In Australia, the volume of fish sold is relatively high and it is of high importance to the industry (Table 7.1). Its risk of spread is therefore high.

Impacts overseas: FishBase regards this species as harmless.

Impacts in Australia: Unknown. None reported to date. Reported to be in two locations (a creek in Brisbane and another near the coast in New South Wales).

4.3 Summary

One point that is immediately obvious is the paucity of impact studies carried out for many of these ornamental species, both in Australia and overseas. Exceptions to this included Mozambique tilapia, goldfish, oriental weatherloach and some of the live-bearing (Poeciliidae) species, which were somewhat more thoroughly studied. However, even for these species, evidence of impacts is often conflicting, based on correlative data or perceptions, or other factors that have been deemed to be potentially responsible for observed patterns in native fish communities. This conclusion is unsurprising, as globally there are very few instances where ecological impacts of alien fish on the receiving fish communities are well understood. This problem has multiple sources, but relates to: 1) inadequate knowledge of implicated species' natural history (habitat requirements, diet, reproduction and behaviour) and their place in their native habitats; 2) similarly inadequate knowledge of the ecosystems into which such species have been inserted as alien species; summing up to a far from adequate understanding of the impacts of the alien species in receiving habitats. All of this generates the attitude, consistent with the precautionary principle, that Simberloff (2003) summed up as "shoot first and then ask questions". Fundamentally, this was his recognition that for the majority of instances, the only safe way of avoiding adverse impacts on an ecosystem by an alien species is to prevent its release and establishment. This sentiment is also expressed in Arthington et al. (1999) and in our view certainly applies to the alien fish species established in Australian waters.

McDowall (2004) concluded for New Zealand that the safest criterion to apply when that country was endeavouring to establish protocols for providing protection from alien aquarium fishes was temperature. In New Zealand, the temperate climate means that the exotic species requiring warm temperatures for reproduction and growth are unlikely to establish, except in isolated habitats below geothermal springs. However, Australia encompasses a much wider range of climate zones and there is a high likelihood that any exotic fish species can find congenial temperatures somewhere in Australian fresh waters. Consequently, all exotic species can potentially establish in the wild somewhere in Australia.

There have been few impact studies focussing on the link between the impact mechanism and impact manifestation for any of the species known to be established in Australian fresh waters. Such mechanistic studies linking cause with effect may help reduce some of the uncertainty surrounding the potential impacts associated with these species because they provide an indication as to whether observed changes in native species composition or relative abundance are due to their displacement, to mortality, reduced reproductive output, or a combination of these. However, it needs to be recognised that determining the actual or potential impacts of any of these species is a major, and therefore costly, exercise, especially given the limited knowledge of the fish involved, of the Australian ecosystems, and the very broad range of habitats available for freshwater fish species to occupy.

Apart from the knowledge gaps relating to evidence of physical impacts on other fish, there are also knowledge gaps relating to the basic biology of many of the listed established ornamental species. As stated earlier, some of these gaps may reflect our inability to secure all relevant information for each species as part of this study. More focussed literature searches on one or a few of these species may well be able to fill some of these gaps without the need for field investigations. We have listed aspects of the biology of each of the listed fish species for which we could find no information. These aspects relate to species' traits linked to potential invasiveness, which is just one side of the potential risk associated with such alien fish. It is therefore, important that these gaps be filled if the risk associated with the 23 established ornamental fish species is to be predicted with any greater certainty.

In this review we have focussed on the impacts of individual fish species, but combinations of exotic species often occur at the same location and their effects may be compounding. For example, McKay (1984) noted that one poeciliid fish appeared to have an appreciable effect on small surface dwelling native species of *Melanotaenia*, *Pseudomugil*, *Craterocephalus* and *Retropinna* in Queensland streams but this affect increased when two or more poeciliids were present. Furthermore, impacts of exotic fish on native species also need to be disentangled from the changes to the physical environment induced by human activities. These two factors (multiple species and physical changes to habitats) pose large challenges to the identification of impacts of ornamental fish in the wild and future research on impacts will need to address such issues.

We could find no information on the hybrid cichlid (*Labeotropheus/Pseudotropheus* cross) so, instead, information is presented for a species from the genus *Labeotropheus*. Given the potential for hybrid fish species to have a different set of environmental tolerances and/or behaviour, all aspects of the biology of this hybrid and impacts associated with it require further investigation. Hybrid vigour can result in fish with better growth performance than pure species, implying greater potential for competition with native species (c.f., Arthington 1991; Mather & Arthington 1991).

While there is general agreement, in term of the risk assessment outputs for the 23 listed ornamental species, between the reports by Arthington et al. (1999) and Bomford & Glover (2004) (Table 3.1) their findings are sometimes at odds with information presented in FishBase, even though these studies, and ours were based on a similar set of criteria relating to potential invasiveness[6]. The basis for risk assessment used by FishBase is unknown, though it appears that reporting of impacts

[6] Even though the assessment frameworks were different.

in at least one country where each of these species is known as an alien might be the primary basis. Of this we cannot be sure. Certainly little information about the nature of associated impacts is given for many species listed by FishBase as being potential pests, and this is quite disconcerting.

We included information about the likelihood of natural dispersal for each of the listed established ornamental fish species and also an assessment of the risk of human-induced spread, but these appraisals are at best conjectural. They are intended to point to where the greatest risks lie. Such information has rarely been included in previous risk assessments. Without such information, risk assessments, based predominantly on traits linked to potential invasiveness, are likely to be highly conservative. Many of the species are likely to have their unassisted dispersal and successful establishment in new waterways limited by minimum winter water temperatures in some parts of Australia, by hydrological conditions, low tolerances of elevated salinity levels, the absence of some preferred spawning requirements, or a combination of these. Clearly, the acquisition of such information is also a high priority for refining estimates of dispersal and impact risk for these species.

We believe that temperature tolerances and preferences may be a good place to start with if for no other reason than that there is some information available associated with the international aquarium trade, and this might provide an ability to estimate the potential geographic ranges of the species involved. Measurement of temperature tolerances is also a relatively straightforward exercise. One caveat to this is that the temperature range required for spawning and egg survival may be higher than that for survival. Both are therefore required to assess the potential geographic range of the species.

Some species, such as the mozambique tilapia, the oriental weatherloach and the oscar are prized angling species in some countries, and could be spread deliberately by coarse fish anglers in Australia. Similarly, the fact that male sailfin mollies exhibit much greater growth of the enlarged dorsal fin in wild habitats could be an incentive to establish feral populations. If this occurs, then assessments of risk based mainly on invasiveness traits could underestimate their potential spread.

There is a wide perception that some exotic species are better than native species at controlling pest populations of mosquitoes and though this generally applies to species of *Gambusia*, some poeciliids are also regarded as having value for this control (e.g., guppies were released for this purpose around Brisbane, pers. comm. A. Arthington). Furthermore, it seems that some adherents of animal rights see it as preferable to release unwanted fish into natural habitats, rather than to destroy them or return them to the ornamental fish trade. For some people, the establishment of wild populations is simply seen as adding to the natural biodiversity of Australian fresh waters. There is clearly an array of factors that we see as having potential to lead to increased ranges for these fish species that broadly cover both natural and human influences, and which need to be managed through public education programmes targeted to change behaviour.

In terms of prioritising the species that require more immediate research attention, our task is somewhat difficult due to the potential for circular processes surrounding choosing species for which there is greater knowledge with respect to risk. If prioritising were to be based purely on known risk, this would favour those species that have already received research attention. Many of these species are listed as noxious or are targeted for risk reduction by existing management plans anyway, despite there being inconclusive evidence with respect to their ecological impacts. On the other hand, there are many species for which there is little information available on any aspect of their potential impacts, and we are in a position of not knowing enough to identify or quantify risk. It is therefore difficult to quantify or rank the risk, and so to know where to start if prioritisation is based on the need to fill as many knowledge gaps as possible. We have acknowledged the disparity between the risk level stated for some species by different workers and, the fact that two of the risk assessments were based mainly on potential invasiveness. Further criteria that we felt might prove helpful in terms of prioritising the risk of species that should receive more immediate future research attention are:

- traits associated with known impact mechanisms such as piscivory and aggression as well as traits associated with indirect effects on fish such as habitat modification;
- traits linked to invasiveness possessed by a particular species including the diversity of habitats it could occupy, its potential geographic range in Australia and propensity to migrate or breed readily;
- whether there is potential for assisted spread by humans;
- reports associating the species with a decline in native fish (or other species).

Clearly, it is difficult to rank the various traits and reports on these species in any objective way and any prioritisation based on an appraisal of all of these factors is necessarily based on what is known about each species, even if this information is meagre. Where there was no information for a particular trait this does not mean that it does not exist, only that its presence or absence has not been reported yet. The absence of information on impacts does not prove that impacts are absent and this, more than anything else, characterises the current status of our knowledge of ornamental fish in the wild in Australia.

The prioritisation of the aquarium fish species is therefore somewhat subjective and likely to be skewed by a lack of information. The rankings provided below in Table

4.12 should therefore be used as a guide only. There is potential for any of the species to be associated with unpredicted impacts should they spread to new locations or should new strains emerge or be introduced into Australia (i.e. the 'residual risk' noted in Arthington et al. (1999)). The ranking therefore establishes the priorities for research to establish the impacts, not the risk of impact itself.

A number of variables were derived from the species assessments carried out in this study to fit under the criteria above and these are listed and explained in Table 4.2 below. Species clearly capable of impacting on a wide range of native fish or already associated with a decline in native fish are colour-coded red. These are regarded as high priorities for research. Other priority species are colour-coded yellow.

Table 4.2: Summary of impact potential based on the reported occurrence of species traits related firstly to direct and indirect effects on other fish populations and secondly to invasive potential based on likely geographic range, habitat range and factors increasing the risk of spread.

Common name	Scientific name	Impact mechanisms					Invasive potential							
		Size[1]	Diet		Behaviour		Likely temperature range[2]		Habitat range[5]			Factors increasing risk of spread		
			Adults	Includes fish	Aggressive	Substrate digger	Preference[3]	Range[4]	Flowing water	Brackish water	Low oxygen	Migratory[6]	Utilisation[7]	Livebearer
Hybrid cichlid	*Labeotropheus-Pseudotropheus*		omnivore		yes									
Jewel cichlid	*Hemichromis bimaculatus*	small	omnivore		yes	yes	warm	narrow		yes		yes		
Victoria Burton's haplochromis	*Haplochromis burtoni*	small					warm	narrow						
Black mangrove cichlid	*Tilapia mariae*	large	herbivore				warm	narrow		yes				
Redbelly tilapia	*Tilapia zillii*	large	herbivore		yes	yes	warm	wide		yes			yes	
Mozambique tilapia	*Oreochromis mossambicus*	large	herbivore		yes	yes	warm	wide		yes	yes		yes	
Oscar	*Astronotus ocellatus*	large	carnivore	yes	yes	yes	warn	narrow			yes		yes	
Three-spot cichlid	*Cichlasoma trimaculatum*	large	carnivore	yes	yes	yes	hot	narrow						
Jack Dempsey	*Cichlasoma octofasciatum*	medium	carnivore	yes	yes		hot	narrow			yes			
Red devil	*Amphilophus labiatus*	medium	carnivore	yes			hot	narrow						
Midas cichlid	*Amphilophus citrinellus*	medium	omnivore	yes			hot	narrow						
Convict cichlid	*Archocentrus nigrofasciatus*	v. small	carnivore	yes	yes		warm	wide	yes					
Blue acara	*Aequidens pulcher*	small	carnivore	yes		yes	warm	narrow	yes				yes	
Green swordtail	*Xiphophorus hellerii*	small	omnivore	yes	yes		hot	narrow	yes	yes	yes			yes
Platy	*Xiphophorus maculatus*	v. small	omnivore				warm	narrow						yes
Sailfin molly	*Poecilia latipinna*	v. small	herbivore				hot	narrow		yes			yes	yes
Guppy	*Poecilia reticulata*	v. small	omnivore				hot	narrow		yes				yes

Table 4.2: (cont.)

Caudo	*Phalloceros caudimaculatus*	v. small	omnivore				warm	narrow		yes			yes	yes
Three-spot gourami	*Trichogaster trichopterus*	small	omnivore				cool	narrow			yes	yes		
Oriental weatherloach	*Misgurnus anguillicaudatus*	medium	omnivore				cool	wide			yes	yes	yes	
Goldfish	*Carassius auratus*	large	omnivore				warm	wide					yes	
Rosy barb	*Puntius conchonius*	medium	omnivore	egg eater			warm	narrow	yes					
White cloud mountain minnow	*Tanichthys albonubes*	v. small	carnivore		yes		warm	narrow	yes					

<u>NOTES</u>

[1]Maximum potential size (large >37 cm, medium 24-25 cm, small 14-16 cm, very small <10cm)

[2]Water temperature range associated with native latitudinal distribution (from FishBase)

[3]Water temperatures expected to be suitable for the species

[4]Width of suitable temperature range (wide = >15 degrees, narrow =<10 degrees)

[5]In general, most species inhabit well oxygenated, still or slow-flowing, freshwaters. This records species with broader habitats

[6]Migratory fish include those that undertake large seasonal migrations to/from spawning areas within river systems

[7]Utility includes use for aquaculture, sports fishery, mosquito control, bait production

5. Impacts associated with the spread of diseases and pathogens

5.1 Introduction

There is no doubt that diseases carried by aquarium fishes represent a significant threat to the ecology and sustainability of Australia's native aquatic fauna. Numerous examples of deleterious impacts from disease agents introduced by aquarium fishes have been recorded both in Australia and overseas (Ashburner 1976; Langdon 1988; Bauer 1991; Stewart 1991; Lumanlan et al. 1992; Arthington & McKenzie 1997; Torchin et al. 2002). Live aquarium fishes are recognised as posing the highest risk group for introducing aquatic animal diseases into Australia, because they are known or potential vectors of numerous diseases of high quarantine significance, are traded widely internationally, and are imported in large numbers into Australia each year (Nunn 1995). Unfortunately, as has been found in other areas of the world (Freyhof & Korte 2005), it appears that the release of imported aquarium fishes into the wild by ill-informed or misguided hobbyists appears inevitable. These actions in turn provide opportunities for exotic disease agents to become established in the Australian environment. Sadly, it is already clear that significant disease agents such as the ciliates *Ichthyophthirius multifiliis*, *Chilodonella hexasticha*, *Trichodina* spp. and *C. cyprini*, and helminths *Gyrodactylus* spp. and *Bothriocephalus acheilognathi* have been spread into native fish populations from exotic fish (both aquarium fish and salmonids) released into the wild, causing significant disease and ecological damage (Ashburner 1976; Langdon 1988; 1990; Rowland & Ingram 1991; Humphrey 1995a; 1995b; Dove et al. 1997; Dove 1998; 2000; Dove & Ernst 1998; Dove & Fletcher 2000; Dove & O'Donoghue 2005). In all cases, these adverse effects are most likely to be permanent and irreversible, and in many cases (e.g., for *I. multifiliis*) will continue to have significant economic consequences for fisheries and fish culturists nationwide.

Another classic example is the spread of goldfish ulcer disease, caused by atypical strains of the exotic bacterium *Aeromonas salmonicida*. This bacterium was first recorded in Australia in 1974 and is thought to have been introduced via infected Japanese goldfish (*Carassius auratus*) (Trust et al. 1980). The disease was first detected at a goldfish farm in South Gippsland, Victoria, in 1975 and spread to other goldfish farms and eventually to populations of feral goldfish (Whittington et al. 1987) and other species, including native fish such as silver perch (*Bidyanus bidyanus*) all through translocation of live, infected goldfish (Langdon 1988; Humphrey & Ashburner 1993). The disease is now considered endemic in southeastern Australia, causing morbidity and mortality in wild, cultured and ornamental fish. Due to the high susceptibility of salmonids to infection, the spread of this disease has resulted in severe restrictions of movements of goldfish into Tasmania, to protect the Atlantic salmon aquaculture industry.

Some authorities even suspect that Australia's first recorded finfish virus, the iridovirus EHNV in redfin perch and various native fishes, may have been introduced by aquarium fish. This is because iridoviruses isolated from ornamental fish (*Poecilia reticulata, Labroides dimidatus*) entering Australia were closely related to EHNV, leading Hedrick and McDowell (1995) to speculate that EHNV may have entered Australia in ornamental fish. More recent work, however, suggests that EHNV may have originated from frogs (Daszak et al. 1999). Redfin that survive epizootics can carry EHNV, which has also been shown to be highly pathogenic to silver perch, mountain galaxias, Macquarie perch and Murray cod (Langdon & Humphrey 1987; Langdon 1990). In the ACT, where mass mortalities of juvenile redfin have been attributed to EHN virus, some authorities consider that this disease has been responsible for major declines in populations of Macquarie perch (Lintermans 1991).

The introduction of these aforementioned disease agents into Australia, and their subsequent establishment in endemic fish species, illustrates the potentially major implications involved with disease transfers from exotic fish. Not only are significant mortality and morbidity experienced by both wild and cultured endemic aquatic animals due to the presence of these disease agents, but also the ongoing economic implications for commercial and recreational fisheries, aquaculture and the ornamental fish industry are significant, and far reaching both spatially and temporally. Even to the casual observer, it is clear there is much potential for irreversible harm to the aquatic environment and fauna of Australia through introduction of disease agents via aquarium fishes, which inevitably, it seems, are eventually released into the wild at some stage by members of the public.

Some may contend that the issue of introduction of disease via imported ornamental fish is not worthy of significant attention as attempts to address the problem are akin to 'shutting the gate after the horse has already bolted'. This is far from the case. Review of the literature shows the problem is clearly ongoing and some may argue, has intensified in recent years due to improvements in transport technology. We must consider recent infections that have spread through fish introductions across the world in the last decade causing serious losses. Good examples include the recent spread of Koi Herpes Virus (KHV) from Japan to Europe and North America (Hedrick et al. 2000), and *Anguillicola crassus* from Asia to Europe, Africa and North America (Peters et al. 1986; Kirk 2003). With further intensive rearing of aquarium fish, without doubt many currently unrecognised infections will become prominent and will be spread through movements of infected fish around the world.

This review was commissioned to examine the impacts of diseases introduced by exotic aquarium fish species that have established wild populations in Australia. Many hundreds of other species of aquarium fish are permitted entry into Australia as part of the approved list of ornamental fish species maintained by the Commonwealth Department of Environment and Water Resources under Part 1 of the live import list established under the Environment Protection and Biodiversity Conservation Act (see http://www.environment.gov.au/biodiversity/trade-use/lists/import/pubs/live-import-list.pdf). Though many of these species harbour disease agents of concern (AQIS 1999), most will not be considered here. Instead, a short list of 23 exotic species, which have already established in Australia (Table 1.2), was considered. A review of the available literature on their diseases was undertaken, with the aim of identifying:

- the potential for introduction of significant disease agents via these fish;
- any gaps in the knowledge of the diseases of these fish;
- criteria for prioritising which species represent the biggest threat; and
- to detail practical approaches towards filling in knowledge gaps and mitigating threats posed by these species.

In the following pages we list the disease agents found in the available scientific literature for the 23 listed species. Despite our best efforts, it is very likely that these lists are not complete, however it is expected the most significant groups of disease agents will be included as significant disease agents, by definition, cause problems which readily become apparent and are therefore more likely to be recorded in the literature. Other problems commonly found during literature searches on disease agents of ornamental fishes include (from Hine and Diggles 2005):

1) The taxonomy of parasites and pathogens of aquarium fish is often uncertain, with species being lumped together one minute, and split the next. Many aquarium fish species originate from developing countries, which generally lack scientific training and expertise on fish diseases. If reports of fish parasites and diseases in developing countries are published, they are often in publications that are very difficult, impossible to obtain, and they are written in the national language. The only method available for access to the abstracts of obscure papers is to use computerised databases, which usually miss out several papers. The abstracts also often lack the details necessary to draw accurate conclusions. A large percentage of such papers are purely taxonomic and give no clues about the pathogenicity of the organism.

In most cases disease agents which are not picked up in electronic database searches will be either purely of taxonomic interest, or not yet well known to science. It is the latter disease agents that generate significant concern when evaluating risks of their introduction, as undoubtedly for many of the 23 fish species listed here, there is little, and sometimes no knowledge of the disease agents present in their countries of origin. It is almost certain that in the next decade or two, some of these new disease agents, most previously unknown to science, will emerge from the ornamental fish trade and cause significant problems in some parts of the globe.

In addition there is a suite of ubiquitous disease agents that are well known to cause disease in aquarium fishes worldwide (e.g., *Ichthyophthirius multifiliis*, *Mycobacterium* spp., *Saprolegnia* spp.). These are listed at the end of the sections devoted to each individual fish species, and it should be considered that each fish species can also be infected by any of the ubiquitous disease agents. *Ichthyophthirius multifiliis*, *Mycobacterium* spp., *Saprolegnia* spp. and indeed most of the other ubiquitous disease agents, have already been recorded from fishes in Australia, many through their introduction in imported ornamental fishes (Humphrey 1995b, AQIS 1999, Evans and Lester 2001).

5.2 Importation of ornamental fish

The importation of ornamental fish is regulated by DEW and AQIS. The DEW live import list (Part 1, Schedule: List of specimens taken to be suitable for live import – Environment Protection and Biodiversity Act 1999) specifies the species of ornamental fish that may be imported based on their potential to damage ecological values. AQIS implements quarantine risk management measures, based on advice from Biosecurity Australia and the outcomes of the Import Risk Analysis on Live Ornamental Finfish (AQIS 1999). Those species on the DEW live import list that were the subject of the Import Risk Analysis on Live Ornamental Finfish (AQIS 1999) or subsequent risk assessments undertaken by Biosecurity Australia are permitted entry by AQIS.

The quarantine risk management measures include pre export (14 days) and post import quarantine periods (7, 14 or 21 days depending on the species). A veterinary health certificate is required from the competent authority for all imports of ornamental fish based on inspection and for goldfish, surveillance and monitoring for specific diseases and treatment with an effective parasiticide prior to export to Australia. Full details of the quarantine risk management measures can be found on the AQIS website at http://www.aqis.gov.au/icon.

Biosecurity Australia is responsible for assessing the quarantine risks associated with the importation of ornamental fish, taking into account Australia's international obligations under the World Trade Organisation (WTO) Agreement on the Application of Sanitary and Phytosanitary Measures (SPS agreement). The quarantine policy for the importation of ornamental finfish may be reviewed when relevant scientific information becomes available that demonstrates that current risk management measures may not be effective. Biosecurity Australia is undertaking a review of the policy in relation to iridoviruses which was announced in March 2005 (Animal Biosecurity Policy Memorandum (ABPM) 2005/01) as a result of research conducted by a student at the University of Sydney which detected gourami iridovirus (GIV) in several species of ornamental gouramis sourced from a pet shop. GIV was considered to be exotic to Australia, however it is unclear if GIV has established in Australia.

5.3 Identification of pathogens associated with aquarium fish species

Relevant literature on the disease agents of the 23 short-listed fish species was obtained through database searches, updating and expanding on previous work done on the species listed in Table 1.2 during the previous reviews of Australian Quarantine Policies and Practices for Aquatic animals and their Products (Humphrey 1995a, 1995b) and the AQIS Import Risk Analysis on live ornamental finfish (AQIS 1999). Web based search engines such as Google, PubMed, IngentaConnect, and Scirius were also utilised in an attempt to generate the most up to date list of disease agents possible for each species. The prefix symbols below are used in the host/parasite lists that follow to provide the reader with a better understanding of the disease status of each fish species and its potential for introduction of exotic disease agents. Absence of a prefix indicates that the pathogen is considered endemic to Australia. Locations for the records are provided when known.

disease agents were not able to be identified to species, but the same genus has been reported in Australia from fish in the wild and/or aquaria or quarantine;
* disease agents previously recorded from Australia in aquaria or quarantine, but may not necessarily occur in wild fish populations;
+ disease agents are considered endemic in the Australian environment, including wild fish populations (endemic disease agents identified using various resources including Humphrey 1995b, AQIS 1999 and other literature).

(a) Hybrid cichlid (*Labeotropheus/Pseudotropheus* cross)

Bacteria
Mycobacterium peregrinum (in *Pseudotropheus*) (Pate et al. 2005)

Algae
Chlorochytrium spp. (in *Pseudotropheus*) algal dermatitis (Yanong et al. 2002)
Scenedesmus spp. (in *Pseudotropheus*) algal dermatitis (Yanong et al. 2002)

(b) Jewel cichlid (*Hemichromis bimaculatus*)

Virus
*+*Lymphocystis* experimental infection, New York (Nigrelli and Ruggieri 1965)

Bacteria
*+*Mycobacterium fortuitum* (Nigrelli & Vogel, 1963)
Mycobacterium ulcerans Ghana (Eddyani et al. 2004)

Metazoa
Gyrodactylus cichlidarum (Paperna 1968)

(c) Victoria Burton's haplochromis (*Haplochromis burtoni*)

No specific disease agents found

(d) Black mangrove cichlid (*Tilapia mariae*)

Myxozoa
Myxobolus nounensis Cameroon (Fomena and Bouix 2000)
Metazoa
Cichlidogyrus testificatus (Justine 2005)
"Heavy intestinal infections by nematode parasites" - Nigeria (King and Etim 2004)

(e) Redbelly tilapia (*Tilapia zillii*)

Virus
Reo Grande cichlid rhabdovirus (Wolf 1988) –isolated in 1 case from a laboratory based peracute disease syndrome.
Protozoa
*+*Chilodonella cyprini* (Paperna 1980)
*+*Chilodonella hexasticha* Israel and Sth Africa, (Paperna and van As 1983)
Cryptosporidium nasorum Egypt (Mahmoud et al. 1998)
#*Eimeria* sp. swimbladder (Landsberg and Paperna 1985)
Eimeria vanasi intestine (Landsberg and Paperna 1987)
Nosemoides tilapeae Africa (Sakiti and Bouix 1987, Lom and Dykova 1992)
*+*Trichodina heterodentata* Philippines (Duncan 1977), Israel (Van As and Basson 1989)
Myxozoa
Myxobolus dahomeyensis (gonad) Benin (Grankoto et al. 2001)
Myxobolus dossoui (gill arch cartilage West Africa) (Grankoto et al. 2001)
Myxobolus heterospora (West Africa) (Gbankoto et al. 2003)
Myxobolus microcapsularis (conjunctive tissue West Africa) (Gbankoto et al. 2001)
Myxobolus zillii (branchial filament, West Africa) (Gbankoto et al. 2001)
Metazoa
Centrocestus sp. (Hine and Diggles 2005)
Cichlidogyrus arthracanthus (Paperna 1996)
Cichlidogyrus aegypticus (Justine 2005)
Cichlidogyrus cubitus (Justine 2005)
Cichlidogyrus digitatus (Justine 2005)
Cichlidogyrus ornatus (Justine 2005)
Cichlidogyrus tiberianus imported from Africa to Israel and established (Hoffman 1970)
Cichlidogyrus yanni (Justine 2005)
#*Clinostomum* sp. (Aloo 2002)
#*Contracaecum* sp. (Aloo 2002)
Gyrodactylus cichlidarum (Paperna 1968)
Haplorchis pumilio (Hine and Diggles 2005)
Polyacanthorhynchus kenyensis (Aloo 2002)

(f) Mozambique tilapia (*Oreochromis mossambicus*)

Virus
*+Bohle iridovirus (BIV) (Ariel and Owens 1997)
#Nodavirus (experimental infection) (Skliris and Richards 1999)

Bacteria
*+*Aeromonas hydrophila* India (Paperna 1980)
*+*Chlamydia* and *rickettsia* (Paperna 1980)
Edwardsiella sp. (Paperna 1996)
*+*Flavobacterium* sp. India (Paperna 1980)
*+*Mycobacterium marinum* - resistant carriers (Wolf and Smith 1999)
*+*Pseudomonas* sp. India (Paperna 1980)
#*Piscirickettsia*-like organism (Mauel et al. 2003)
**Staphylococcus faecalis* (Bunkley-Williams and Williams 1994)
*+*Streptococcus iniae* India (Mukhi et al. 2001)
*+*Vibrio alginolyticus* (Bunkley-Williams and Williams 1994)
*+*Vibrio vulnificus* (Bunkley-Williams and Williams 1994)

Fungi
*+*Aphanomyces invadans* (Thailand) (Tonguthai 1985; Lio-Po et al. 2000)
#*Branchiomyces*-like fungus Israel (Paperna and Smirnova 1997)

Protozoa
*+*Amyloodinium ocellatum* (Paperna 1996)
Eimeria vanasi intestine (Landsberg and Paperna 1987)
Goussia cichlidarum experimental infection (Paperna and Landsberg 1985)
*+*Ichthyophthirius multifiliis* – this fish species was more resistant than other species (Subasinghe and Sommerville 1990)
Trypanosoma choudhuryi India (Mandal 1977)
*+*Trichodina heterodentata* (Dove and O'Donoghue 2005)
Trichodina mutabilis (Hine and Diggles 2005)
Metazoa
Argulus sp. Bangaledesh (Arthur and Ahmed 2002)
*+*Bothriocephalus acheilognathi* Southern Africa (Paperna 1996)
Centrocestus formosanus (Hine and Diggles 2005)
Cichlidogyrus sp. Imported from Africa to USA and established (Hoffman 1970)
Cichlidogyrus sclerosis Philippines (Humphrey 1995a)
Cichlidogyrus tilapiae (Bunkley-Williams and Williams 1994)
Euclinostomum heterostomum (Donges 1974)
Enterogyrus cichlidarum Africa, Paperna 1996, introduced into SE Asia (Natividad et al.1986)
Gyrodactylus cichlidarum (Bunkley-Williams and Williams 1994)
Gnathostoma binucleatum Mexico (Almeyda-Artigas 1991)
Neobenedenia melleni (Bunkley-Williams and Williams 1994)
Scutogyrus chikhii Republic of Congo (Pariselle and Euzet 2003)
Ophiotaenia sp. (Bunkley-Williams and Williams 1994)
Ophiovalipora minuta (Bunkley-Williams and Williams 1994)

(g) Oscar (*Astronotus ocellatus*)

Virus
Iridovirus (Yanong and Terrell 2003)

Bacteria
*+*Aeromonas hydrophila* (Soltani et al. 1998)
*+*Pseudomonas fluorescens* (Humphrey 1995b)
**Salmonella typhimurium* Sweden (Lundborg and Robertsson 1978, Hongslo et al. 1987)

Protozoa
**Hexamita* sp. (hole in the head) (Humphrey 1995b)
*+*Ichthyobodo necator* (Humphrey 1995b)

Metazoa
Ancyrocephalus sp. (Bunkley-Williams and Williams 1994)
Argulus japonicus (Bunkley-Williams and Williams 1994)
#*Dactylogyrus* sp. (Humphrey 1995b)
Goezia sp.(Humphrey 1995b)
Gussevia asota Korea (Kritsky et al. 1989, Kim et al. 2002a)
#*Procamallanus* sp. (Humphrey 1995b)

(h) Three-spot cichlid (*Cichlasoma trimaculatum*)

Metazoa
Gnathostoma binucleatum zoonotic nematode in skeletal muscle Mexico (Martinez-Salazar and León-Règagnon 2005).
Sciadicleithrum mexicanum Guatamala (Mendoza-Franco et al. 2000)

(i) Jack Dempsey (*Cichlasoma octofasciatum*)

Metazoa
Ascocotyle nunezae (metacercaria) Mexico (Scholz et al. 1997)
Bothriocephalus musculosus (Baer 1937, Scholz et al. 1996)
Crassicutis cichlasomae Mexico (Scholz et al. 1995, Salgado-Maldonado et al. 2005)
Capillaria pterophylli Czechoslovakia (Moravec and Gut 1982)
Genarchella isabellae Mexico (Scholz et al. 1995)
Neoechinorhynchus golvani Mexico (Salgado-Maldonado et al. 2005)
Oligogonotylus manteri Mexico (Salgado-Maldonado et al. 2005)
Sciadicleithrum mexicanum Mexico (Mendoza-Franco et al. 1999)
Sciadicleithrum bravohollisae Mexico (Salgado-Maldonado et al. 2005)
Spiroxys sp. Mexico (Salgado-Maldonado et al. 2005)

(j) **Red devil (*Amphilophus labiatus*)**

Metazoa
Sciadicleithrum nicaraguense Nicaragua (Vidal-Martinez et al. 2001)

(k) **Midas cichlid (*Amphilophus citrinellus*)**

Metazoa
#*Procamallanus* sp. (Martinez et al. 2002)
Sciadicleithrum nicaraguense Nicaragua (Vidal-Martinez et al. 2001).

(l) **Convict cichlid (*Archocentrus nigrofasciatus*)**

Metazoa
Sciadicleithrum bicuense Nicaragua, (Vidal-Martinez et al. 2001)
Sciadicleithrum meekii Nicaragua (Mendoza-Franco et al. 2003)
Spiroxys sp. Mexico (Martinez et al. 2002)

(m) **Blue acara (*Aequidens pulcher*)**

Virus
*+*Lymphocystis* Trinidad (found by a tropical fish dealer in New York, Nigrelli and Ruggieri 1965).

(n) **Green swordtail (*Xiphophorus hellerii*)**

Virus
Platyfish virus-like particles (Wolf 1988)
Iridovirus (Paperna et al. 2001)

Protozoa
*+*Trichodina heterodentata* Brisbane (Dove 2000, Dove and O'Donoghue 2005)
#*Trichodina* sp. Sri Lanka (Thilakaratne et al. 2003)

Metazoa
Ascocotyle mcintoshi Mexico (Salgado-Maldonado et al. 2005)
Ascocotyle nana Mexico (Salgado-Maldonado et al. 2005)
Ascocotyle tenuicollis Mexico (Salgado-Maldonado et al. 2005)
*+*Bothriocephalus acheilognathi* Hawaii (Vincent and Font 2003)
**Camallanus cotti* Singapore, Hawaii (Vincent and Font 2003)
Centrocestus formosanus Mexico (Salgado-Maldonado et al. 2005)
Clinostomum complanatum Mexico (Salgado-Maldonado et al. 2005)
#*Dactylogyrus* sp. Sri Lanka (Thilakaratne et al. 2003)
#*Ergasilus* sp. Sri Lanka (Thilakaratne et al. 2003)
*+*Gyrodactylus bullatarudis* Queensland (Dove and Ernst 1998)
Gyrodactylus rasini Czech Republic (Lucky 1973)
#*Gyrodactylus* sp. Sri Lanka (Thilakaratne et al. 2003)
Mexiconema cichlasomae Mexico (Montoya-Mendoza et al. 2004)

Pygidiopsis pindoramensis Mexico (Salgado-Maldonado et al. 2005) *Saccocoelioides sogandaresi* Mexico (Salgado-Maldonado et al. 2005) *Rhipidocotyle* sp. Mexico (Salgado-Maldonado et al. 2005) *Spiroxys* sp. Mexico (Salgado-Maldonado et al. 2005) *Uvulifer ambloplitis* Mexico (Salgado-Maldonado et al. 2005) Urocleidoides vaginoclaustrum India (Jogunoori et al. 2004)

(o) Platy (*Xiphophorus maculatus*)

Virus

Platyfish virus-like particles (Wolf 1988)

Protozoa

*+*Trichodina heterodentata* Brisbane (Dove 2000, Dove and O'Donoghue 2005)
#*Trichodina* sp. Sri Lanka (Thilakaratne et al. 2003)

Metazoa

Argulus sp. Sri Lanka (Thilakaratne et al. 2003)
*+*Bothriocephalus acheilognathi* Australia –post-quarantine (Evans and Lester 2001)
**Camallanus cotti* Singapore (Levsen and Berland 2002)
**Centrocestus formosanus* Australia –post-quarantine (Evans and Lester 2001)
#*Dactylogyrus* sp. Sri Lanka (Thilakaratne et al. 2003)
#*Trichodina* sp. Sri Lanka (Thilakaratne et al. 2003)
#*Ergasilus* sp. Sri Lanka (Thilakaratne et al. 2003)
*+*Gyrodactylus bullatarudis* Korea (Kim et al. 2002a)
#*Gyrodactylus* sp. Sri Lanka (Thilakaratne et al. 2003)
*+*Prototransversotrema steeri* Brisbane (Dove 2000)
**Urocleidoides reticulatus* Australia –post-quarantine (Evans and Lester 2001)

(p) Sailfin molly (*Poecilia latipinna*)

Protozoa

#*Ambiphyra* sp. Texas (Tobler et al. 2005)
*+*Ichthyophthirius multifiliis* (McCallum 1986)
#*Ichthyobodo* sp. Texas (Tobler et al. 2005)
#*Oodinium* sp. Texas (Tobler et al. 2005)
#*Trichodina* sp. Texas (Tobler et al. 2005)

Metazoa

Acanthocephalus cf. alabamensis Texas (Tobler et al. 2005)
Ascocotyle leighi USA (Burton 1956)
**Camallanus cotti* (Langdon 1988)
#*Dactylogyrus* sp. Texas (Tobler et al. 2005)
#*Lernaea* sp. Texas (Tobler et al. 2005)
#*Postodiplostomum minimum* Texas (Tobler et al. 2005)
Saccocoelioides sogandaresi USA (Lumsden 1963)
Transversotrema patialense (Whitfield et al. 1986)
Unidentified nematode Texas (Tobler et al. 2005)
Uvulifer ambloplitis Texas (Tobler et al. 2005)

(q) Guppy (_Poecilia reticulata_)

Virus

Iridovirus (Hedrick and McDowall 1995)
[#]Nodavirus Singapore (Hegde et al. 2003)
*Reo-like virus Australia-quarantine (Humphrey 1995a)

Protozoa

*[+]_Tetrahymena pyriformis_ (experimental) (Ponpornpisit et al. 2000)
*_Tetrahymena corlissi_ Australia –post-quarantine (Evans and Lester 2001), Korea (Kim et al. 2002a)
[#]_Tetrahymena_ sp. Sri Lanka (Thilakaratne et al. 2003), Israel (Pimenta Leibowitz et al. 2005)
*[+]_Trichodina heterodentata_ (Dove and O'Donoghue 2005)
*[+]_Trichodina acuta_ (Dove and O'Donoghue 2005)
[#]_Trichodina_ sp. Sri Lanka (Thilakaratne et al. 2003)

Myxozoa

Myxobolus nuevoleonensis Mexico (Segovia-Salinas et al. 1991)

Metazoa

*[+]_Bothriocephalus acheilognathi_ Australia –post-quarantine (Evans and Lester 2001)
*_Camallanus cotti_ Australia –post-quarantine (Evans and Lester 2001), Korea (Kim et al. 2002a)
[#]_Capillaria_ sp. Sri Lanka (Thilakaratne et al. 2003)
*_Centrocestus formosanus_ Australia –post-quarantine (Evans and Lester 2001)
[#]_Dactylogyrus_ sp. Sri Lanka (Thilakaratne et al. 2003)
Diplostomum pseudospathaceum (Hine and Diggles 2005)
[#]_Ergasilus_ sp. Sri Lanka (Thilakaratne et al. 2003)
*[+]_Gyrodactylus bullatarudis_ Queensland (Dove and Ernst 1998)
Gyrodactylus turnbulli (Harris 1986)
Ñapillaria tomentose Russia (Skiba 1998)
Saccocoelioides tarpazensis Venezuela (Diaz and Gonzalez 1990)
Transversotrema patialense (Whitfield et al. 1986)
*_Urocleidoides reticulatus_ Australia –post-quarantine (Evans and Lester 2001)

(r) Caudo (_Phalloceros caudimaculatus_)

No specific disease agents found

(s) Three-spot gourami (_Trichogaster trichopterus_)

Virus

IPNV (Humphrey 1995b)
[*]Gourami iridovirus (Fraser et al. 1993, Go et al. 2005)
Iridovirus (Paperna et al. 2001)
*[+]Lymphocystis (Durham and Anderson 1981)

Bacteria

*[+]_Aeromonas hydrophila_ (Fock et al. 2001)
*[+]_Edwardsiella tarda_ (Dixon and Contreras 1992, Ling et al. 2001)

#*Mycobacterium* sp. (Santacana et al. 1982)
#*Nocardia* sp. (Paperna 1996)
*+*Vibrio anguillarum* (Fang et al. 2000)

Fungi

*+*Aphanomyces invadans* Thailand (Tonguthai 1985, Catap and Munday 1999)

Protozoa

Goussia trichogasteri (Szekely and Molnar 1992, Kim and Paperna 1993)
Trichodina heterodentata Philippines (Duncan 1977)
Valkampfia debilis (Lom and Dykova 1992)

Metazoa

Camallanus anabantis (Nimai 1999)
Transversotrema patialense Malaysia (Seng 1988)

(t) Oriental weatherloach (*Misgurnus anguillicaudatus*)

Virus

IPNV (covert infection Chou et al. 1993, OIE 2003)

Bacteria

*+*Flavobacterium columnare* (AQIS 1999)

Protozoa

**Piscinoodinium pillularis* (Langdon 1988)
*+*Trichodina heterodentata* Taiwan (Basson and Van As 1994)

Metazoa

Centrocestus complanatum (Lo et al. 1981, Paperna 1996).
*+*Clinostomum complanatum* Japan (Langdon 1988)
#*Diplostomum* sp. Japan (Miyamoto 1987)
Echinostoma cinetorchis Korea (Seo et al. 1984).
Echinostoma hortense Korea, Japan (Chai et al. 1985, Miyamoto 1987, Ryang 1990)
Gnathostoma nipponicum Japan, China (Ando et al. 1988, Sohn et al. 1993)
*+*Gyrodactylus macracanthus* ACT (Ergens 1975, Dove and Ernst 1998)
Gyrodactylus micracanthus (Ergens 1975)
Gyrodactylus monstruosus USSR (Gusev 1955)
Gyrodactylus misgurni China (Ling 1962)
Gyrodactylus strelkovi USSR (Ergens and Danilov 1980)
Massaliatrema misgurni China (Ohyama et al. 2001)
Metagonimus sp. Japan (Miyamoto 1987)
Paracaryophyllaeus gotoi Japan (Scholtz et al. 2001)

(u) Goldfish (*Carassius auratus*)

Virus

Black moor herpesvirus (Humphrey 1995b)
Goldfish iridovirus (Wolf 1988)
*+Haematopoietic necrosis herpesvirus of goldfish (CyHV-2)(Jung and Miyazaki 1995, Stephens et al. 2004)

*[+]Herpes-like virus (Humphrey 1995b)
Infectious spleen and kidney necrosis virus (He et al. 2002)
IPNV (Adair and Ferguson 1981)
Spring Viraemia of Carp (OIE 2003)

Bacteria

*[+]_Aeromonas salmonicida_ atypical (Humphrey 1995b)
*[+]_Edwardsiella tarda_ (Humphrey 1995b)
*[+]_Mycobacterium_ sp. (Humphrey 1995b)
*[+]_Pseudomonas fluorescens_ (Humphrey 1995b)
*[+]_Streptococcus_ sp. (Humphrey 1995b)
*[+]_Vibrio cholerae_ (non-O1) (Humphrey 1995b)
*[+]_Yersinia ruckeri_ (Humphrey 1995b)

Fungi

*[+]_Aphanomyces invadans_ (Chinabut and Roberts 1999)
Pythium undulatum (Alderman 1982)

Protozoa

Acanthamoeba sohi Korea on gills, pathogenic to mice (Im and Shin 2003)
*[+]_Chilodonella cyprini_ (Humphrey 1995b)
*[+]_Chilodonella hexasticha_ (Humphrey 1995b)
*[+]_Cryptobia_ sp. (Humphrey 1995b)
*[+]_Eimeria_ sp. (Humphrey 1995b)
Goussia carpelli (Lom and Dykova 1992)
[#]_Ichthyobodo_ sp. (Humphrey 1995b, Thilakaratne et al. 2003)
[#]_Pleistophora_ sp. (Lom and Dykova 1992)
*[+]Systemic amoebiasis (_Dermocystidium_-like) (Voelker et al. 1977, Humphrey 1995b)
[#]_Tetrahymena_ sp. Sri Lanka (Thilakaratne et al. 2003)
*[+]_Trichodina reticulata_ (Dove and O'Donoghue 2005)
[#]_Trichodina_ sp. (Humphrey 1995b, Thilakaratne et al. 2003)
Trypanoplasma borreli (Lom and Dykova 1992)
Trypanosoma carassii (Lom and Dykova 1992)
Trypanosoma danilewskyi (Islam and Woo 1991)
Vannella platypodia (Dykova et al. 1996)

Myxozoa

*[+]_Hoferellus carassii_ (_Mitaspora cyprini_) (Lom and Dykova 1992)
Myxobolus carassii (Lom and Dykova 1992)
Myxobolus cultus Japan (Yokoyama et al. 1995)
Myxobolus diversus China, Hungary (Molnar and Szekely 2003)
[#]_Myxobolus_ sp. (Humphrey 1995b)
Sphaerospora molnari (Lom and Dykova 1992)
Sphaerospora renicola (Lom and Dykova 1992)
[#]_Sphaerospora_ sp. (Humphrey 1995b)

Metazoa

Argulus sp. Sri Lanka (Thilakaratne et al. 2003)
*[+]_Bothriocephalus acheilognathi_ (Mitchell and Hoffman 1980, Dove and Fletcher 2000)
Centrocestus sp. Sri Lanka (Thilakaratne et al. 2003)
*[+]_Dactylogyrus anchoratus_ (Dove and Ernst 1998)

Dactylogyrus dulkeiti Czech Republic (Simkova et al. 2004) *Dactylogyrus extensus* Sri Lanka (Thilakaratne et al. 2003) *Dactylogyrus formosus* Czech Republic (Simkova et al. 2004) *Dactylogyrus inexpectatus* Czech Republic (Simkova et al. 2004) *Dactylogyrus intermedius* Czech Republic (Simkova et al. 2004) *Dactylogyrus vastator* (Wootton 1989, Thilakaratne et al. 2003) #*Dactylogyrus* sp. (Humphrey 1995b, Thilakaratne et al. 2003) *Gyrodactylus carassii* (Malmerg 1957) *+*Gyrodactylus kobayashii* (Jones et al. 1997) *Gyrodactylus shulmani* China (Ling 1962) *Gyrodactylus sprostonae* China (Ling 1962) #*Gyrodactylus* sp. (Humphrey 1995b, Thilakaratne et al. 2003) *+*Lernaea cyprinacea* (syn. *L. elegans*) (Humphrey 1995b) #*Lernaea* sp. (Humphrey 1995b, Thilakaratne et al. 2003) Philometroides sanguinea (Bauer 1991).

(v) Rosy barb (*Puntius conchonius*)

Virus *Rosy Barb virus Australia - quarantine (Langdon 1990) **Bacteria** *+*Aeromonas hydrophila* India (Devashish et al. 1999) **Edwardsiella ictaluri* Australia – quarantine (Humphrey et al. 1986) #*Streptococcus* sp. Humphrey (1995b) **Fungi** *+*Aphanomyces invadans* Thailand (Tonguthai 1985, Roberts et al. 1986) **Protozoa** **Piscinoodinium pillulare* Malaysia (Shaharom-Harrison et al. 1990) **Metazoa** #*Gyrodactylus* sp. Philippines (Lumanlan et al. 1992) *Procamallanus spiculogubernaculus* (Hine and Diggles 2005) *Pseudocapillaria margolisi* India (De and Maity 1996)

(w) White cloud mountain minnow (*Tanichthys albonubes*)

Bacteria #*Streptococcus* sp. Canada (Ferguson et al. 1994) **Protozoa** #*Trichodina* sp. Philippines (Lumanlan et al. 1992) **Metazoa** *Transversotrema patialense* (Whitfield et al. 1986)

5.4 Knowledge gaps

Despite little evidence of targeted research into the parasitology and disease agents of most of the species listed, published records of parasites and disease agents were found for 21 of the 23 species. Twenty of those 21 species were shown to harbour at least one disease agent exotic to Australia. In addition, as the review progressed it became clear that each species could harbour a number of ubiquitous disease agents, which are well known and commonly described from aquarium fishes worldwide. Many of these ubiquitous agents cause significant disease. However, all these ubiquitous disease agents (with the exception of *Argulus foliaceus*) have already been recorded in Australia, many probably being introduced through importation of exotic fish (Langdon 1988; Humphrey 1995a,b).

The organisms that are ubiquitous and common to many species of fish are listed in Table 5.1.

Table 5.1: Ubiquitous pathogens found in Australian freshwater fish.

Viruses	*+Lymphocystis Virus
Bacteria	*+*Aeromonas hydrophila* *+#*Flexibacter* spp *+#*Flavobacterium* spp. *+#*Mycobacterium* spp. *+#*Vibrio* spp.
Fungi	*+#*Aphanomyces* spp. *+#*Branchiomyces* spp. *+#*Pythium* spp. *+#*Saprolegnia* spp.
Protozoa	*+#*Amyloodinium* spp. *+*Ichthyobodo necator* *+*Ichthyophthirius multifiliis* *+#*Oodinium* spp. *+#*Piscinoodinium* spp. *+#*Tetrahymena* spp. *+#*Trichodina* spp. *+#*Trichodinella* spp.
Metazoa	*+#*Dactylogyrus* spp. *+#*Gyrodactylus* spp. Argulus foliaceus

Major knowledge gaps that became evident during the course of the review included the following:

1) There was a lack of knowledge of the parasitology and disease agents of aquarium fishes in their countries of origin. For most of the listed species, there was little evidence that there had ever been a thorough examination for

parasites undertaken by suitably qualified persons in their countries of origin. Indeed, for two of the species listed (Victoria Burton's haplochromis and one spot live bearer) no record of any specific disease agents could be found in the literature. This suggests that few, if any, studies of the disease status of these species has been done in their countries of origin. Furthermore, evidence of surveillance for viruses and bacteria, appeared virtually non-existent in most cases. This is not surprising, as although most of the serious diseases of finfish are viral or bacterial in nature, the resources and expertise required to conduct effective bacteriological and virological studies are unavailable to most of the poorly developed countries currently supplying ornamental fishes to the trade. In other countries, the expertise and resources to study these disease agents in fish may be available, but disease outbreaks due to bacteria or viruses are generally not investigated until there are significant economic or public health implications. This is also the case in Australia. It is not surprising, therefore, that Evans & Lester (2001) concluded that there are potentially large numbers of unknown and undescribed parasites being transported worldwide with the aquarium fish trade. Their observation may well hold for viruses and bacterial disease agents too.

2) Although Australia has the expertise to identify and monitor fish disease outbreaks in fish, the resources for extensive pro-active monitoring are lacking. Furthermore, there is a lack of knowledge of the parasitology and disease agents of Australian native fishes. Although a large amount of work has been done in this area (see review by Humphrey (1995a,b)), knowledge of the parasitology and disease agents, which naturally occur in Australian fishes, remains incomplete. For example, Dove & O'Donoghue (2005) studied the trichodinid ciliate ectoparasites of introduced and native fishes to determine which species have been introduced with exotic fishes and to determine the extent to which these species have crossed into native fish populations. They found 21 putative species of *Trichodina* in 33 species of fish examined, and used a simple formula to estimate that the biodiversity of these parasites in Australian freshwater fish may approach 150 species. Their paper outlined how incomplete knowledge of natural parasite fauna of our freshwater fishes becomes a problem whenever new parasites and disease agents are discovered. The key question of whether the agent occurs naturally in wild fish populations, or has been introduced via exotic species, often cannot be answered with any certainty due to a lack of baseline information. Furthermore, as pointed out by Dove and O'Donoghue (2005), thorough taxonomic studies are required to determine the true extent of species diversity of parasites of native fish populations, so that phenomena such as host switching between native and introduced species can be better recognised where and when it occurs.

3) There was extremely poor knowledge of the parasites and disease agents of introduced fishes in Australia and their impact on native fish populations. The parasite fauna and disease agents of introduced fishes in Australia has been virtually unstudied, except for the work done by Langdon (1988, 1990) and more recently, Evans & Lester (2001) and Dove and co-authors (i.e., Dove 1998; Dove & Ernst 1998; Dove & Fletcher 2000; Dove & O'Donoghue 2005). The paper by Rowland & Ingram (1991) discussed a number of parasites found on native fish species, including Murray cod, golden perch and silver perch, and suggested that at least one of the parasites found on native fishes (*Lernaea* sp.) was introduced by common carp. However generally speaking there appears to be a significant knowledge gap surrounding the parasites and disease agents of introduced fishes in Australia, which is surprising given the value of commercial and recreational fisheries and the current rapid expansion of aquaculture in this country.

5.5 Prioritisation of species in terms of their risk to fish health

Humphrey (1995a, b) undertook a most thorough assessment of the quarantine threat of diseases of aquatic animals, including aquarium fishes. His work clearly showed that aquarium fishes are recognised vectors of exotic diseases of high quarantine significance, but indicated that the aquarium fish industry, apart from quarantine provisions, is essentially unregulated with regard to movements of fish and potentially, their diseases around the country (Humphrey 1995a).

The risks of introduction of disease agents with aquarium fish is clear; however the extent of their threat to the health status of native fish and other aquatic organisms will vary depending on the type of disease agent, and its host. For example, many parasites have high host specificity (e.g., monogeneans), while others tend to have complex multi host lifecycles (e.g., digeneans, cestodes, nematodes), characteristics that tend to reduce the risk of transfer of these parasites to native species. Obviously for these species, the risk of transfer tends to increase when exotic and native fish species are closely related. Thus the high level of endemicity of Australian fishes may assist in reduction of risks of transfer of some exotic metazoan parasites from introduced fishes (Dove & Ernst 1998). However, many invasive metazoan parasite species have low host specificity (e.g., *Bothriocephalus acheilognathi*, *Camallanus cotti*), and viruses, bacteria and protozoa seldom exhibit high host specificity (Dove & O'Donoghue 2005). Thus the introduction of these types of disease agents could potentially threaten the health of native fishes and other aquatic fauna, regardless of the identity of the exotic host vector.

Humphrey (1995a) recommended that aquarium fishes be divided into low and high-risk groups on the basis of their potential threat to Australia's aquatic environment and quarantine/trade status. Goldfish (*C. auratus*), guppies (including *Poecilia reticulata*) and gouramis (including *Trichogaster trichopterus*) were recognised as special cases for immediate inclusion in the high risk category, as all three species are recognised vectors of exotic diseases of high quarantine importance, all are farmed in their countries of origin in open systems with direct access to natural waters, and large numbers are currently imported into Australia (Humphrey 1995a, b). The literature review done here reinforces the position of Humphrey (1995a) by showing that goldfish, guppies and gouramis harbour a number of very pathogenic disease agents which would almost certainly cause significant and irreversible damage to Australia's indigenous fish fauna, fisheries and aquaculture industries and adversely affect the countries trading status if they were introduced into wild fish populations. For example, three-spot gourami (*Trichogaster trichopterus*) is a known carrier of IPNV, which can cause epizootic mortalities in salmonids and marine fishes, and also iridoviruses which potentially threaten both native fish and amphibians (see below).

Recent disease outbreaks in farmed Murray cod (Lancaster et al. 2003) were caused by new iridoviruses, which are suspected to have been introduced into Australia by aquarium fish. There is currently no direct evidence of introduction of these iridoviruses into wild fish populations via aquarium fishes: however Go et al. (2005) showed that several species of diseased gouramis (subfamily Trichogastrinae of the family Osphronemidae) sampled from pet shops in Sydney harboured exotic strains of iridovirus (namely a tropivirus related to dwarf gourami iridovirus (DGIV) which exhibited over 99.6% sequence homology with the iridovirus isolated from the diseased cultured Murray Cod (Go et al. 2005)). In experimental cohabitation trials, the virus isolated from diseased gouramis was then transmitted to captive Murray cod, causing mortalities of 36.6% within 28 days (Go et al. 2005). Furthermore, intraperitoneal injection of organ filtrates from infected gouramis caused 96.6% mortality of Murray cod within 28 days (Go et al. 2005). This information strongly suggests that gouramis can act as vectors of iridoviruses pathogenic to native fish. Hence populations of gouramis established in the wild may be reservoirs for exotic iridoviruses which can cause mortalities in native fishes which are both economically important, and threatened in many parts of their range by habitat destruction and river flow alterations. Furthermore, some iridoviruses carried by amphibians (members of the Ranavirus group) are known to cause disease in fish (Moody & Owens 1994; Cullen et al. 1995). The close relatedness of tropiviruses and ranaviruses suggests that the converse may also occur (Daszak et al. 1999), suggesting viruses carried by gouramis may cause disease in frogs, many species of which are already endangered in Australia. This information has lead to Biosecurity Australia undertaking a re-assessment of the quarantine risk associated with imports of freshwater ornamental finfish with respect to iridoviruses (Biosecurity Australia 2005).

The present review confirmed that goldfish are host to a very large number of disease agents. Many of these have already been identified in goldfish in Australia, and some, such as *Aeromonas salmonicida*, have already been transferred to native fish species (Humphrey 1995b). However a number of significant viral, protozoan, myxozoan and metazoan parasites of goldfish remain exotic to Australia at this time. In particular the viruses such as IPNV and Spring Viraemia of Carp are listed by the OIE and their introduction would have significant negative ramifications for Australia's trading status as well as potentially causing significant morbidity and mortality in native fishes and salmonids. For these reasons, goldfish must certainly be considered amongst the highest risk ornamental fish species imported into Australia. In fact, because of the high disease risk posed by this species, some countries have banned the importation of goldfish (Hine & Diggles 2005), instead relying on production by local producers to meet demand by hobbyists for this species.

It has been suggested by some authorities that guppies (*Poecilia reticulata*) may have introduced the iridovirus EHNV which causes disease and mortalities in redfin perch and various native fishes. This is because iridoviruses isolated from *P. reticulata* entering Australia were closely related to EHNV (Hedrick and McDowell 1995). The fact that guppies and other poeciliids (*Xiphophorus*) harbour a range of iridoviruses and nodaviruses suggests they deserve to be included in the highest risk category. Furthermore, the non-host specific, pathogenic, Asian nematode *Camallanus cotti,* has spread in Southeast Asia, Europe, North America, Hawaii and Australia with the trade in guppies (see Levsen & Berland 2002; Levsen & Jakobsen 2002). Examination of guppies imported into Korea showed 14.4% prevalence (Kim et al. 2002a), and in those entering Australia the prevalence was 48% (Evans & Lester 2001). *Camallanus cotti* caused 30% mortalities following introduction into an ornamental fish farm in Korea, where it infected 71% of the cultured fishes (Kim et al. 2002b). *Camallanus cotti* normally uses planktonic copepods as intermediate hosts, but if they are not present, it can infect directly, fish-to-fish (Levsen & Jakobsen 2002). After guppies were introduced into Hawaii for mosquito control, *C. cotti* jumped host into 5 native fish species, including an eleotrid (*Eleotris sandwicensis*) (see Font & Tate 1994; Font 1998), and many of Australia's freshwater gobies and gudgeons are members of the family eleotridae (Allen 1989). Langdon (1988) reported *C. cotti* as being present and causing disease in captive populations of sailfin mollies (*P. latipinna*), but it remains unclear whether this parasite has established in wild populations of poeciliids in Australia. The work done by Dove et al. (1997), Dove (1998), Dove & Fletcher (2000) suggests that it may have not yet done so, which is good news for many species of smaller native fishes, at least until more work is done on their parasite faunas, which may prove otherwise.

A significant parasite commonly found in poeciliids is the digenean *Centrocestus formosanus*, a gill trematode, which causes significant losses in juvenile tropical fish culture (Blazer & Gratzek 1985; Vogelbein & Overstreet 1988). This parasite occurred in *Poecilia* spp. and *Xiphophorus* spp. in Australia at prevalences up to 100% after clearing quarantine (Evans & Lester 2001). This parasite is virtually non host specific for fish second intermediate hosts, but is more specific for the first intermediate host, which is usually the snail *Melanoides tuberculata* (see Vogelbein and Overstreet 1988; Mitchell et al. 2005). *M. tuberculata*, a member of the family thiaridae, has been confirmed as being introduced into tropical Australia (http://www.environment.gov.au/ssd/new/watersnail.html), and there are also a number of other species of native thiarid snails which might also act as intermediate hosts for this parasite in Australia, which uses water birds (herons and egrets) and mammals as the final host (Mitchell et al. 2005). This complex lifecycle could therefore be completed in tropical wetlands in the northern parts of Australia. The introduction and establishment of this parasite in Australia would pose threats to the health of a wide variety of juvenile native fishes, as it has done in the USA where it threatens a number of endangered fish species (Mitchell et al. 2005).

This review therefore prioritises all three species groups described above as representing the highest disease risk to native aquatic fauna. Goldfish (*Carassius auratus*), gouramis (including *Trichogaster trichopterus*), and poeciliids (*Poecilia* spp, *Xiphophorus* spp.) imported from overseas all host significant exotic viruses and/or parasites which could adversely affect native fauna. The fact that all three of these high risk fish species have been released and have developed natural self sustaining populations reinforces the conclusion that there is also a similarly high risk of introduction and establishment of the diseases carried by these species.

Of the remaining species, we consider a second group should also be prioritised as representing a lesser, though still significant risk. This medium risk group includes fish which host one or two significant exotic disease agents with low host specificity, and/or parasites of zoonotic importance. They include Mozambique tilapia (*Oreochromis mossambicus*), oriental weatherloach (*Misgurnus anguillicaudatus*), and rosy barb (*Puntius conchonius*).

Mozambique tilapia is host to a variety of disease agents of concern, including iridoviruses and nodaviruses, *C. formosanus*, *B. acheilognathi*, and a number of exotic protozoan and metazoan parasites. To date there has been little study of the disease agents present in populations of Mozambique tilapia established in Australia, and therefore the full extent of the impact of the establishment of wild populations of these fish remains undetermined.

The oriental weatherloach, is a known covert carrier of IPNV (OIE 2003). The establishment of this species has already introduced the monogenean *Gyrodactylus macracanthus* but that parasite has high host specificity and is unlikely to infect native fishes (Dove and Ernst 1998). However, the oriental weatherloach hosts a number of zoonotic helminth parasites (*Echinostoma hortense*, *Gnathostoma nipponicum*) the larvae of which infect a wide range of fishes and which cause disease in mammals and humans in Asia (Miyamoto 1987; Ryang 1990; Sohn et al. 1993). These parasites occur at high prevalences in wild weatherloach and probably also in cultured loach in their countries of origin. The presence of these parasites would not be detected in quarantine. The larval parasites survive a long time in the host and adult parasites could reduce the health of native wildlife and even humans if they ate undercooked weatherloach. The decision to stop importation of weatherloach into Australia (Dove & Ernst 1998) would therefore appear a good one, when the disease agents this species could potentially introduce into the country are considered.

Rosy barb was the final species identified as posing a medium risk. This was because it has been recognised as harbouring a birna-like virus called rosy barb virus which has previously been isolated from this species in quarantine (Langdon 1990). This species is also a known covert carrier of *Edwardsiella ictaluri*, an exotic bacterium listed by the OIE (2003) because it causes epizootic mortalities in catfishes and a variety of other fish species. The introduction of this bacterium would have significant negative ramifications for Australia's' trading status as well as potentially causing significant morbidity and mortality in salmonids and a variety of native fish species, many of which are likely to be susceptible to the pathogen. Closely related species in the genus *Barbus* are also known to carry other viruses, including IPNV (Ortega et al. 1993a, 1993b).

These examples show that diseases carried by the high and medium risk aquarium fish species listed above represent a significant threat to the ecology and sustainability of Australia's native aquatic fauna, in some cases potentially native waterbirds and mammals, and in the case of zoonotic agents, even to human health. The remaining species are considered to represent a lower risk of disease introduction to native aquatic fauna. Mostly this lower risk status has been based on the fact that most of the records found for the remaining species are either ubiquitous disease agents which already occur in Australia, or are parasites with complex lifecycles (nematodes, digeneans, cestodes) and/or high host specificity (monogeneans). Both these latter factors tend to reduce the risk of introduction of disease agents into native fish populations (Dove 1998; 2000; Dove & Ernst 1998). In particular, the establishment of exotic monogenean populations on Australian native fishes via host-switching is considered less likely than for other parasitic groups due to the generally high host-specificity of monogeneans, combined with the phylogenetic dissimilarity of native and exotic fishes (Dove & Ernst 1998). However, caution must be emphasised before discounting the disease threats posed by establishment of these lower risk fish species. This is because in most cases little if any research has been conducted on their disease status either in Australia or their countries of origin. Many fish could well harbour new disease agents that are presently not known to science (Evans & Lester 2001), and it is therefore impossible to estimate the full extent of the risk posed by these unknown disease agents, though they undoubtedly do pose some level of risk (Gaughan 2002).

5.6 Summary and recommendations

Many examples of deleterious impacts from disease agents introduced by exotic aquarium fishes have been recorded both in Australia and overseas (Asburner 1976;

Whittington et al. 1987; Langdon 1988; Bauer 1991; Stewart 1991; Lumanlan et al. 1992; Arthington & McKenzie 1997; AQUIS 1999; Torchin et al. 2002; Chong & Whittington 2005). For example, in Australia, significant disease agents such as the bacterium *Aeromonas salmonicida*, ciliates *Ichthyophthirius multifiliis*, *Trichodina* spp., *Chilodonella hexasticha*, and *C. cyprini*, and helminths *Gyrodactylus* spp. and *Bothriocephalus acheilognathi* have all been spread into native fish populations from exotic fish released into the wild, most causing significant disease and ecological damage (Asburner 1976; Whittington et al. 1987; Langdon 1988, 1990; Roland & Ingram 1991; Humphrey & Ashburner 1993; Humphrey 1995a, 1995b; Dove et al. 1997; Dove & Ernst 1998; Dove 1998, 2000; Dove & Fletcher 2000; Dove & O'Donoghue 2005). In addition, some authorities suspect that Australia's first recorded finfish virus, the iridovirus EHNV in redfin perch and various native fishes, may have been introduced by aquarium fish. Recent iridovirus outbreaks in farmed Murray cod are caused by another iridovirus, also probably introduced into Australia by aquarium fish. These examples show that diseases carried by aquarium fishes represent a significant threat to the ecology and sustainability of Australia's native aquatic fauna. The introduction of diseases via aquarium fish also has the potential to result in significant, irreversible, and economically detrimental effects on Australia's fisheries and aquaculture industries.

A review of the disease agents recorded in the scientific literature from a short list of 23 exotic species that have established in Australia was undertaken with the aim of identifying; their potential for introduction of significant disease agents, gaps in the knowledge of their diseases, criteria for prioritising which species represent the biggest threat, and to detail practical approaches towards filling in knowledge gaps and mitigating threats posed by these species.

Despite little evidence of research into the parasitology and disease agents of most of the species listed, published records of parasites and disease agents were found for 21 of the 23 species. Twenty of those 21 species harboured disease agents exotic to Australia. In addition, each species can harbour a number of ubiquitous disease agents, many of which can cause significant disease. However, nearly all these ubiquitous disease agents have already been recorded in Australia, many undoubtedly already present via importation with exotic fish.

Major knowledge gaps which were evident included lack of knowledge of the parasitology and other disease agents (particularly viruses and bacteria) of aquarium fishes in their countries of origin, lack of knowledge of the parasitology and endemic disease agents of Australian native fishes, and extremely poor knowledge of the parasites and disease agents of introduced fishes in Australia and their impact on native fish populations. The latter appears to be a particularly significant knowledge gap given the value of commercial and recreational fisheries and the current rapid expansion of aquaculture in this country.

By reviewing the number and types of disease agents carried by each species on the shortlist, the following species were prioritised as representing the highest disease risk to native aquatic fauna: goldfish (*Carassius auratus*), gouramis (including *Trichogaster trichopterus*), and poeciliids (*Poecilia* spp., *Xiphophorus* spp.). All three host significant exotic viruses and/or large numbers of exotic parasites, which could adversely affect native aquatic fauna.

Other species were also prioritised as representing a lesser, though still significant risk. These medium risk species include Mozambique tilapia (*Oreochromis mossambicus*), oriental weatherloach (*Misgurnus anguillicaudatus*), and rosy barb (*Barbus conchonius*). All three of these species host significant exotic disease agents with low host specificity, and/or other parasites of zoonotic importance which could pose a threat not only to native fish, but also to the health of humans, waterbirds and/or other warm blooded native terrestrial animals.

The remaining species on the list were considered to represent a lower risk of disease introduction to native fauna, due to the increased likelihood of them harbouring parasites with high host specificity and/or complex lifecycles. Both these factors tend to reduce the risk of introduction and establishment of disease agents into native fish populations. However, caution must be emphasised before discounting the threat posed by establishment of these species, as in most cases little if any research has been conducted on their disease status either in Australia or their countries of origin, and even less on their potential to infect native fish species.

Possible ways of filling the current knowledge gaps were considered to include:

- increased surveillance of the parasites and disease agents of aquarium fish traded internationally;
- increased surveillance and taxonomic study of the parasites and disease agents of Australian native fishes in the wild; and
- increased surveillance, taxonomic and epidemiological study of the parasites and disease agents of introduced fishes.

Practical ways of mitigating the disease threat posed by these species were considered to include:

- increased public education to reduce the frequency of hobbyists releasing aquarium fish into the wild; and
- fostering studies to scan both native and exotic fish populations in the wild to determine the presence/absence and distributional ranges of exotic parasites and disease organisms;

- providing Biosecurity Australia with relevant new scientific information to support a review of current quarantine risk management measures when this information demonstrates that current risk management measures may not be effective.

There is much potential for the aquarium fish industry to play an important role in helping to implement these recommendations, particularly in relation to public awareness of the risks associated with the release of aquarium fish into the wild. Considerable research and surveillance and monitoring of diseases and parasites of ornamental fish is needed and this will require adequate funding to achieve. Finding the funds for such research will be the key to achieving this goal.

The industry is at the 'coalface' when it comes to detecting diseases in newly introduced fish and has practical experience in preventing and handling disease. It is in the interests of fish importers, breeders and distributers to minimise disease risk to their stocks. Therefore the industry could play an important co-operative role by helping in the set-up and implementation of disease monitoring systems and by encouraging the breeding within Australia of exotic species with potentially high disease laodings. A pro-active role in disease monitoring and prevention would reduce the overall risk to the industry's economic base while providing an indirect benefit to native fish and other Australian freshwater resources by reducing the risk of disease transfer.

6. Impacts associated with genetic changes

6.1 Introduction

The genetic threats posed to native fauna by the introduction of exotic aquarium fish is discussed in the following section. There can be little doubt that hybridisation, introgression and the breakdown of species boundaries is a significant threat to biodiversity and native fish species worldwide (Weigel et al. 2002; Arthington 1991). The main genetic threats to native fish fauna are likely to be: 1) hybridisation and introgression, 2) problems associated with small populations due to deleterious ecological interactions and disease, 3) hybridisation and 4) impacts from genetically modified fish. The latter is not considered here as, at present, this technology is experimental and genetically modified fish have not been released into the wild. Please note, where possible we have used fish as examples to illustrate points in the following discussion.

Hybridisation historically has been defined in several distinct ways. Classically, supporters of the biological species definition (Mayr 1963) suggest that hybridisation is the crossing of two distinct species in which resulting offspring are not evolutionarily viable (sterile). From an evolutionary biology standpoint, distinct lineages of species are an intrinsic and important level of biological diversity. Therefore, a better definition would be the crossing of evolutionarily distinct populations. Consequently, this review uses the definition of Arnold (1997) where “natural hybridisation involves successful mating between individuals from two populations, which are distinguishable on the basis of one or more heritable characteristics”. However, for this review, the primary goal is to discuss the effects of species level hybridisation between endemic and introduced taxa.

Introgression is the movement of genetic material between separate species/ populations through hybridisation and backcrossing between fertile hybrids and either parental line (Stebbins 1959). Though hybridisation can and does commonly occur (Arnold 1997), introgression can only occur if hybrids are fertile and genetically compatible with either parental species/population (Dowling & Childs 1992).

6.2 Isolating mechanisms

To better understand the threat posed by the hybridisation of endemic and introduced fish fauna, we need to understand the mechanisms that increase both the likelihood of inter-species crosses and those isolating structures that prevent them.

In his review on the subject, Templeton (1981) suggested that the primary isolating mechanisms that prevent inter-species hybridisation can be split into three general categories, namely 1) pre-mating isolation, 2) post-mating isolation and 3) post-zygotic isolation. Pre-mating isolation barriers consist of phenotypic, temporal, ecological and ethological differences between species. Post-mating barriers include differing reproductive mechanisms and gametic incompatibilities, whereas post-zygotic isolation will manifest as non-viability of F_1 (first generation) progeny, F_1 sterility and F_1 backcross breakdown.

Many sympatric species (species with overlapping distributions) have evolved distinct niches and breeding regimes specific to their environment. In fish, these various breeding systems are thought to be intrinsically linked to environmental cues such as ambient temperature, photoperiod and riverine flow. Intrinsic differences in these reproductive traits are a result of phenotypic, temporal, ecological and ethological preferences.

(A) Pre-mating isolation mechanisms

Phenotypic characters*:* Though phenotypic characters are the result of various interactions between genome and environment (natural selection), the development of distinct morphological characters for sexual selection is similarly important. The simplest method higher organisms retain to distinguish themselves from other species is through distinct morphological characters (Arnold 1997). Predominantly these characters are size, body shape, appendage shape, colour patterns and location of characters (Hubbs 1955). Generally, the closer the evolutionary relationship, the more morphologically similar species will appear to be. A well known exception is convergent evolution, where species appear to share a similar evolutionary lineage based on appearance, but have merely arrived at a similar morphotype based on chance and similar selective pressures, not by shared ancestry. At the crudest level, large differences in size and overall body shape will determine species boundaries. However, once large scale differences are accounted for, it is in the detail that will distinguish species. For example, colour choice has been shown to be the dominant factor in mate choice in tropical hamlets (*Hypoplectrus:* Serranidae), where observations in the wild suggest that spawning is almost exclusively (~95%) between individuals of the same colour pattern (Fischer 1980). Colour pattern distinction is also known for butterfly fish (*Chaetodon*) (Palumbi 1994). These small but distinct differences are an effective mechanism to maintain reproductive isolation and evolutionary distinction.

Temporal isolation: For external, mass spawners like fish, temporal spawning asynchrony will play a significant role in separating gametes in time and space (Palumbi 1994). Temporal differences in mating systems are likely to be driven by environmental variability over time. Generally, organisms reproduce when particular resources and conditions become available. In many freshwater native fish these differences are likely to be access to certain flow conditions, temperatures, water quality and food. For example, Murray cod are known to build nests and spawn in complex habitat where the large adhesive eggs can be guarded against predation by a parent. This takes place over spring and early summer at a water temperature ranging from 15°C to 23°C (Harris & Rowland 1996). The congeneric trout cod however, spawns earlier in the season at a slightly lower temperature (Cadwallader & Lawrence 1990). These preferences are likely to keep both congenerics separate during the spawning period. However, these two species have been reported to hybridise when confined in time and space in artificial habitats such as Prospect Reservoir (S. Rowland pers. comm.).

Ecological isolation: One of the most common inhibitors to cross-species mating is spatial dissimilarities in distribution. Species that have allopatric (non-overlapping) distributions are unlikely to come into contact with congenerics and therefore cannot reproduce with them. For sympatric species, a spatial difference in spawning habitat is a primary isolation mechanism (Arnold 1997). Australian native fish have very particular and often distinct requirements for spawning. For example, yellowfin bream (*Acanthopagrus australis*) spawn in river mouths and surf zones, whereas the sympatric black bream (*Acanthopagrus butcheri*) spawns well inside river systems. Only when this spatial isolation is interrupted do hybrids occur. Rowland (1984) found hybrids between both species in intermittently landlocked coastal lakes, where both were locked together in space and time. Golden perch (*Macquaria ambigua*) is known to spawn large planktonic eggs during peak flow events when the lower floodplain is breached and inundated inducing a successional phytoplankton/ zooplankton bloom (Cadwallader & Lawrence 1990). Blooms are likely to provide a greater range of zooplankton sizes for larval fish to graze, as opposed to static plankton populations which tend to be much more uniform in size. Macquarie perch (*Macquaria australasica*) on the other hand are believed to prefer montane higher energy streams dominated by boulders, pebbles and gravel, where the slightly adhesive eggs sink among the substrate (Harris & Rowland 1996). These life history differences are very effective at isolating both species reproductively.

Ethological isolation: Behavioural dissimilarities in mating between closely related species are likely to be a very strong isolating mechanism. Many organisms have developed elaborate mating displays distinct to their individual species. For example, the sympatric satin (*Ptilonorhynchus violaceus*) and regent bowerbirds (*Sericulus chysocephalus*) both build elaborate bowers (freestanding upright ground nests) in which they place brightly coloured ornaments to attract mates. However, each species builds its bower in a slightly different way and decorate them with different coloured ornaments. The quality of the nest, and the type, colour and quantity of the ornaments on display are all integral in the reproductive success of individuals (Simpson & Day 1993). Poorly built or furbished nests are likely to result in no mating or offspring and therefore would provide quite a significant isolating mechanism.

Distinct behavioural characteristics have been documented for fiddler crabs (genus Uca), which engage in elaborate courtship displays in which males wave and rap their claw to attract partners (Palumbi 1994). Other small crab species do not have the same courtship display, and therefore are unlikely to be attracted to fiddler crabs for mating.

It should be noted that a native fish example was not used in this section due to the paucity of data for pre-mating behaviour in Australian fish fauna. In most cases either data were available for one sympatric species or no closely related taxa coexist. For example, pre-spawning courtship has been observed for eastern freshwater cod (*Muccullochella ikei*) but there are no data for Mary River cod (*M. peelii mariensis*), Murray cod (M. peelii peelii), or trout cod (*M. macquariensis*) (G. Butler pers. comm.). Indeed, the nesting behaviour and parental care have still not been witnessed in the wild for these last three species (S. Rowland pers. comm.).

(B) Post-mating isolation mechanisms

Many groups of aquatic taxa, such as fish, sponges, corals, bivalves, ascidians and echinoderms have no courtship behaviour and, being external spawners, they release their gametes *en masse*. For some groups, corals are a good example, the group spawning takes place under certain environmental conditions, and many species have synchronised gametic release. As a result of this mass en masse spawning system, many groups have developed post-mating isolating mechanisms. The actual mechanics of reproduction and fertilisation are complex and are known to vary between taxonomic groups (Rundle 2002). The primary differences are likely to be gametic incompatibilities that have built up as species diverged through time, and isolation. Some species have developed self-compatibility mechanisms that actively reject gametes if they are incompatible (Kao & Huang 1994). The number and compatibility of chromosomes are known to vary between groups, as are the size of germ-line cells like sperm (Wade & Johnstone 1994). Such differences between taxa are likely to pose a significant barrier to reproduction. Additionally, as these isolated species/populations move through evolutionary time and space they are likely to develop larger reproductive incompatibilities. Post-mating isolation, observed as sperm/egg incompatibilities, have been reported in aquatic invertebrates, such as sea urchins (Palumbi & Metz 1991; Metz et al. 1994) and polychaetes (Marsden 1992). In the case of sea urchins, crossing trials were conducted between taxa with only slight morphological differentiation and that are similar enough to have once been classified as different morphotypes of the same species. Molecular evidence suggests they most likely shared a direct common ancestor. Despite these similarities, strong incompatibilities during sperm-egg attachment prohibits fertilization. In such cases, species boundaries are not crossed, reinforcing these boundaries.

(C) Post-zygotic isolating mechanisms

Even when reproduction occurs and offspring are produced, isolating mechanisms may still play a significant role in maintaining species' distinctions. It is quite common for F_1 progeny to be sterile, halting backcrosses with either parental line. In some cases, even in F_1 progeny are fertile, backcrosses with parental species may be halted by incompatibilities between the hybrid and parent (Rhymer & Simberloff 1996). In both these situations, there will be little or no introgression of genetic material between either parental species. For example, 97% of hybrids detected between the introduced brook trout (*Salvelinus fontinalis*) and bull trout (*S. confluentus*) are F_1 crosses (Leary et al. 1993), suggesting that some form of isolating mechanism is keeping the F_1 crosses from mating with either parental line. The meagre amount of parental backcrossing is likely to produce very low levels of introgression between parental species. In some cases, exchange of genetic material may be unidirectional as with Apache trout (*Oncorhynchus gilae apache*), where genes from translocated rainbow trout (*Oncorhynchus mykiss*) have introgressed into Apache trout genomes, but the reverse has not occurred (Dowling & Childs 1992).

Even if the mechanics of reproduction can be overcome, divergent selection on the offspring can lead to isolation. Intermediate phenotypes may be less well adapted to a particular environment than either parental species, with no intermediate niches to exploit. For example, divergent selection was shown to play a central role in the evolution of post-zygotic isolation between benthic and limnetic forms in sympatric sticklebacks. Intermediates do not perform as well as parental species in each habitat and are selected against, reinforcing species boundaries (Rundle 2002).

6.3 Hybridisation between native and introduced fish

The biological species definition that delineates species as being reproductively isolated from all other species (Mayr 1963), is not perfect and indeed species, especially plants (Gillet 1972; Levin et al. 1996) and fish (Hubbs 1955; Avise & Saunders 1984; Rubidge & Taylor, 2005) hybridise continually. Indeed hybridisation is likely to be an important mechanism in the evolutionary process. The major determinant for the likelihood of hybridisation and introgression between species will be their evolutionary relatedness over all other factors, for it is incompatibilities at the chromosomal and genetic level that will prevent the production of offspring. Fortuitously, Australia's fish fauna is highly endemic and does not contain major groups common to most other large land masses. Australia has no native members of the families Poeciliidae, Cichlidae, Cobitidae, Osphronemidae or Cyprinidae to which all the exotic aquarium fish belong. Therefore the genetic threats to native fish via hybridisation, introgression, and the dilution of species boundaries must be considered negligible, despite there being little research on the topic.

6.4 Genetic implications of demographic contraction

Interactions between native and exotic species are likely to be negative in many ways (Costedoat et al. 2004; Gurevitch & Padilla 2004). These negative interactions in some situations have the potential to reduce abundance within or fragment native populations (Wayne et al. 1992). For example, *Gambusia* may help to fragment populations of native fish by reducing or eliminating native competitors in some sensitive areas (Moore, unpublished data). If the reduction in number is significant enough, genetic factors are likely to affect the fitness and persistence of those populations.

Populations that contract in size or become fragmented may suffer from inbreeding depression, and the loss of allelic diversity and heterozygosity. Large stable populations are expected to be at equilibrium between the loss of genetic variation through genetic drift and the creation of new diversity through natural mutation events (Hartl & Clark 1997). Populations that decrease in size below this equilibrium state are likely to lose genetic variation over time. This loss can be in the form of a decrease in the number of alleles (variations at a particular gene locus) or in heterozygosity. Both forms of genetic variation are important for population and individual health. Heterozygosity is most likely to affect individual fitness in the short term, whereas allelic diversity is likely to give a population adaptive potential to cope with stochastic environmental events and new predators, competitors, parasites and diseases over evolutionary timescales (Soulè 1980).

These natural population bottlenecks also increase the likelihood of a population suffering inbreeding and the resultant deleterious consequences of inbreeding depression. The negative effects of inbreeding are well documented (Ralls & Ballou 1983; Gall 1987) and include decreases in individual and Darwinian fitness (Wright, 1977) and increases in deformed offspring (Kincaid 1976a; Kincard 1976b) and extinction probability (Saccheri et al. 1998). This reduction in overall phenotypic fitness is believed to be a result of an increase in the expression of recessive deleterious alleles (Hartl & Clark 1997).

The general trend of decreasing population fitness can be reversed if the population can recover demographically to large sizes in time. The effects of the bottleneck will depend on the severity, length and nature of the bottleneck (Frankel & Soulè 1981).

6.5 Hybridisation between introduced fish

Though hybridisation between current introduced and native fish taxa is very unlikely, hybridisation within introduced taxa is quite probable and could create hybrids with greater environmental tolerances and adaptive potential for colonising new niches. An understanding of the role of hybridisation in evolution may well be critical for managing exotic fishes in the future.

There can be little doubt that hybridisation contributes to the evolutionary process. From the neo-Darwinian viewpoint, several key processes drive evolutionary change in populations including mutation, recombination, drift, natural selection (both at the biochemical and ecological level), sexual selection and environment. Hybridisation and introgression are likely to affect populations in several important ways. The most commonly recognised affect of hybridisation is the production of infertile offspring due to post-zygotic isolating mechanisms and reduced recruitment as a result of gametic incompatibilities or the breakdown of stable embryological pathways (Rhymer & Simberloff 1996; Arnold 1997). However, hybridisation within certain groups is a regular occurrence and commonly produces viable offspring, especially in plants (Stebbins 1959; Gillett 1972; Levin et al. 1996) and fish (Hubbs 1955; Avise & Saunders 1984; Rowland 1984; Campton 1987, Baker et al. 2002; Rubidge & Taylor 2005; Buonaccorsi et al. 2005) and other vertebrates (Ferris et al. 1983; Lehman et al. 1991; Wayne et al. 1992). In fact fish show some of the highest levels of hybridisation in vertebrates (Verspoor & Hammar, 1991). The resultant introgression of genetic material between two parental groups can have both positive and negative affects on their evolution (Stebbins 1959).

Positive effects of hybridisation for exotic species: The process of introgression of new genetic material to populations that are either small, or have gone through a recent bottleneck or founder event, can be very positive. It is expected that small populations lose genetic variation through genetic drift faster than it can be maintained through mutation. Thus, most populations that have survived severe demographic bottlenecks or founder events have lost a significant portion of their allelic diversity (Moore 2000). This genetic diversity is essential in the evolutionary process as it provides adaptive potential for populations through evolutionary time (Frankel & Soulè 1981). A loss in adaptive potential increases the risk of extinction (Soulè 1980). The resultant increase in Darwinian fitness in the F_1 generation as a result of hybridisation is known as heterosis or hybrid vigour. It is likely that the more depauperate the gene pool, the greater the increase in vigour.

Given that all introduced aquarium fish are likely to have been through at least one significant founder event (though presumably multiple demographic bottlenecks), they may well benefit from the introgression of new genetic material. In these cases the progeny are likely to show higher levels of fitness and adaptability than their parents, with the ability to invade new ecological niches (Lewontin & Birch 1966). The production of novel hybrid genotypes could therefore result in adaptive evolution and the displacement of parental species by their offspring (Arnold 1997).

Therefore the crossing of two groups of exotic fish may result in a more vigorous pest species that out-competes its parents and other native fish. A case in point would be the crossing of European carp (*Cyprinus carpio*) varieties to produce the Boolara strain, which is now dominant in Australia (Arthington 1991). The Boolara strain (named after Boolara in South-eastern Victoria where it was first released) has been far more invasive than two previous varieties released in Prospect Reservoir and the Murrumbidgee Irrigation area in New South Wales (Shearer & Mulvey 1978). Despite the long-term persistence of both these populations (introduced by 1908 though may have been as early as in the 1860), it was the liberation of the Boolara strain in the 1970's that resulted in the large-scale spread of the species throughout Australia (Morison & Hume 1989). The original two stockings appear to be quite benign in comparison to the hybrid form. The incorporation of new genetic material may help explain why a species that has gone through several demographic bottlenecks is such an aggressive and adaptive coloniser. Founder populations are thought unlikely to be as adaptive as we have seen with carp, though cane toads and *Gambusia* are other examples where founder populations are aggressive adaptors. It must be noted, that the impact of bottlenecks is a function of the severity and length of the contraction. In the case of species that have significantly increased in abundance such as carp, *Gambusia* and cane toads would be acquiring new genetic material through mutation under new selective pressures much faster than populations that stay small.

Negative affects of hybridisation for exotic species: The deleterious effects of hybridisation are complex and likely to affect populations and species differently in space and time. Identified problems include reductions in reproductive output, increases in non-viable hybrids, reduction of fitness in intermediate forms, loss of species distinction for parental forms, and reduction or loss of parental forms through competition with differently adapted offspring.

The production of offspring via the reproductive coalescence of two individuals will not always lead to introgression. Commonly, the offspring are reproductively unfit (sterile). In many species hybrid swarms can be dominated by sterile F_1 hybrids, with no backcrossing with either parental stock. Hubbs (1955) describes swarms of sterile F_1's making up 95% of the base population of sunfish. Such hybrids may have been known to aggressively dominate parental species and defend spawning habitat with greater vigour than parental lines (Hubbs 1955). Any subsequent spawning between sterile hybrids and parental species is likely to be wasted reproductive effort, which can be catastrophic in bottlenecked populations. These interactions are likely to have a detrimental effect on the parental species, especially if the parental stock is small and under stress from other threats.

Hybridisation is likely to lead to intermediate forms in many instances. These intermediate forms can be less fit than ancestral forms as a result of being less well-adapted to the local environment. This reduction in fitness in intermediate forms is a result of outbreeding depression. Outbreeding depression can include both the loss of locally adapted traits or the breakdown of co-adapted gene complexes. Forms of outbreeding depression can be seen in anadromous salmonid fishes (Gilk et al. 2004). Hybridisation within the group has had a detrimental affect on spawning timing, ability to find suitable spawning habitat, orientation of newly emerged fry and overall reproductive fitness (Rhymer & Simberloff 1996). Granath et al. (2004) found higher survival rates in control lines of Alaskan coho salmon (*Oncorhynchus kisutch*) than hybrids formed by crossing geographically separate populations of the species. Such changes can erode fitness and weaken a population and in some cases be catastrophic if the selective pressure on the trait is strong enough. For example, the Tatra mountain Ibex (*Capra ibex ibex*) population in Czechoslovakia was eliminated as a result of crossing with a subspecies from Turkey. The introduced population was intrinsically linked to its own locally adapted traits (a warmer drier climate). The resulting hybrids rutted in autumn instead of winter and gave birth in mid-winter, resulting in the local extinction of the species (Templeton 1997).

6.6 Likelihood of hybridisation between introduced fish fauna

The 30 species of introduced aquarium fish that have established within Australia (Table 1.1) represent five distinct families that are non-indigenous to the Australian landscape. Hybridisation and introgression within each family is likely and in some cases has already occurred. The consequences can be quite significant, but due to a paucity of research in the area, is something that will all too likely go undetermined.

Cichlidae: The Cairns population of Mozambique tilapia (*Oreochromis mossambicus)* was thought to be a hybrid cross with *O. hornorum* and possibly *O. niloticus* (Blühdorn et al. 1990). Mather & Arthington (1991) later found that the tilapia in the Cairns region comprise two morphs with one being a strain of *Tilapia mariae* and the other a hybrid between *Oreochromis massambicus* and another *Oreochromis* species (viz., *O. niloticus*, *O. aureas*, or *O. honorum*). The potential for further hybridisation in introduced populations of these species is quite high if the current trend of liberation continues. No data are presently available on whether the hybrid form of this species is outperforming other strains in Australia, but Mather & Arthington (1991) noted that hybrid vigour and enhanced reproductive potential can result in hybrids outperforming pure species. Mozambique tilapia are known to be a ready coloniser and have the potential to extend their current distribution, especially if the introgression of new genetic material provides greater adaptive potential (Arthington 1991). Evidence has also emerged that an intermediate form of *Labeotropheus sp*. and *Pseudotropheus sp.* has been found in the thermal discharge of the Hazelwood power station in the La Trobe River in Victoria. This location may prove to be a hotspot of cichlid hybridisation, with one African species and an African hybrid form (*Tilapia mariae* & *Labeotropheus sp*. and *Pseudotropheus sp.* cross), one Central American (*Amphilophus labiatus*) and two South American species (*Archocentrus nigrofasciatus* and *Aequidens pulcher*) occurring in artificial sympatry. Similarly, the Ross River in North Queensland contains cichlids. The evidence of hybridisation between two genera *Labeotropheus* and *Pseudotropheus* may add some weight to this hypothesis.

Osphronemidae: There is only one species (three-spot gourami *Trichogaster trichopterus*) from the Family Osphronemidae in Australia, which occurs in the Ross and Burdekin Rivers and Sheepstation Creek in North Queensland. To date there is no evidence of hybrid forms or alternate strains within Australia, with the species central to a single region in Queensland. Therefore there is a very low threat of hybridisation with other species or strains at this stage.

Cobitidae: Due to taxonomic uncertainties with classification, it is unclear whether there are one or two species of weatherloach in Australia and hybridisation is known to occur in the family (Morishima et al. 2002). A molecular systematic study would be required to ascertain what species are currently present and if a threat exists. *Misgurnus anguillicaudatus* has 50 diploid chromosomes and *M. mizolepis* 48 (Koster et al. 2002), which may lead to post-mating isolation.

Cyprinidae: There are presently six introduced members of the family Cyprinidae that have established self-reproducing populations in Australia. These include European carp (*Cyprinus carpio*), goldfish (*Carassius auratus*), white cloud mountain minnow (*Tanichthys albonubes*), rosy barb (*Puntius conchonius*), roach (*Rutilus rutilus*) and tench (*Tinca tinca*). Hybridisation has been reported between goldfish (*Carassius auratus*) and European carp (*Cyprinus carpio*) throughout Victoria including drainages of the Murray (Hume 1983). Hybrids between Yanco strain carp and goldfish have been detected in the Murrumbidgee Irrigation Area in New South Wales (Shearer & Mulley 1983) as have intraspecific hybrids of Yanco and Boolara strain carp (Mulley & Shearer 1980). Indeed the Boolara strain of European carp, which is the dominant form of carp in Australia, is believed to be a hybrid strain between at least two varieties (Arthington 1991). There is also strong international evidence that carp commonly hybridise (Costedoat et al. 2005). The evidence that this group can and does hybridise suggests that we may well see more examples as research is directed into this area and the spread of the group continues.

Poeciliidae: There are now six known species belonging to the family poeciliidae (from Central and South America) established in Australia. These comprise the sailfin molly (*Poecilia latipinna*), guppy (*Poecilia reticulata*), green swordtail (*Xiphophorus hellerii*), platy (*Xiphophorus maculatus*), one-spot livebearer (*Phalloceros caudimaculatus*) and mosquitofish (*Gambusia holbrooki*). Poeciliids are known to hybridise in the wild (Hubbs 1955; Scribner 1993; Rosenthal et al. 2003) and in captivity (Scribner & Avise 1994; Lima 1998; Scribner et al. 1999; Mitchell et al. 2004), indeed the Amazon molly (*Poecilia formosa*) is a recognised hybrid species (Hubbs 1955; Schartl et al. 1995; Lamatsch et al. 2002; Dries 2003; Tiedemann et al. 2005; Lambert 2005). Within the Australian context there remains little evidence of multiple strains or hybridisation within the family, though morphological and genetic differences have been found across the range for *G. holbrooki* (Arthington 1991). Additional research is required to determine if hybridisation is occurring.

6.7 Summary of the genetic implications of aquarium fish

Hybridisation, introgression and the breakdown of species boundaries pose a significant risk to biodiversity throughout the world. The old paradigms of the biological species being reproductively isolated from each other does not hold under empirical analysis. Particular groups, such as fish, readily hybridise, indeed hybridisation and introgression appear to be an intrinsic part of the evolutionary process.

The threats of hybridisation, introgression and the breakdown of species boundaries posed by exotic aquarium fish on native fish should be seen as negligible at present. This argument is derived from the fact that Australia's fish fauna is highly endemic and does not support the major fish families represented by exotic aquarium fish

(Arthington 1991), As has been described, the differences between these introduced and native taxa are very likely to be sufficient to prevent any form of species crossing.

However, the genetic threats posed by exotic aquarium species are likely to be as a result of decreases in abundance and the fragmentation of populations due to negative ecological and disease interactions. These effects are likely to have some deleterious consequences for genetic diversity, as well as individual and population health. The deleterious consequences of small population size are likely to be increases in inbreeding and the loss of fitness associated with inbreeding depression and the loss of allelic diversity and heterozygosity. Those species or populations likely to suffer the greatest genetically will be those that are reduced to the smallest population size.

Hybridisation within exotic aquarium fish has already happened to some degree and has the potential to happen in the future. Hybridisation within exotic fish fauna raises the threat of producing hybrids with greater fitness and increased adaptability and which can then expand into new ecological niches as has occurred with carp in Australia. Other than eradication, there appears very little action that can be taken to remove or decrease this threat.

The paucity of research into basic biological information on reproduction, systematics, population genetics and impacts of introduced taxa in Australia suggests that research priorities need to be focused on these issues if we are to move forward. It is likely that this information may prove useful in the control of these taxa in Australia.

7. Economic and social values of aquarium fish in Australia

7.1 Economic value of the aquarium industry

Background and approach: The aquarium fish industry in Australia comprises imports of aquarium fish species, breeding (domestic production) of aquarium fish, sale of fish through the wholesale and retail markets, commercial aquariums that are open to the public, and sale of food and accessories that are necessary for keeping aquarium fish. The value of all of these activities taken together represents the gross value of the aquarium fish industry.

The data available on these aspects of the aquarium fish industry are limited. Aquarium fish are usually retailed to the public through pet shops and the retailers are represented by an association of pet shop owners. Pet shops sell many more products than aquarium fish and accessories, although some pet shops might specialise in aquarium fish. Using total sales from pet shops, if such data were available, would give a misleading impression of the value of the aquarium fish industry. Values that are indicative of the minimum gross value of the industry provide a less confused measure of the value of the industry.

The total economic impact of the aquarium industry in Australia has never been evaluated. There are, however various measures that speak to the value of an industry, such as volume of production, international trade levels, the turnover of the retail or wholesale sector and the level of employment either directly or indirectly resulting from the industry. This chapter provides a description of the aquarium industry in Australia, its size and scope, in order to provide an indication of the importance of the industry that may be affected by management, control and eradication options put forward.

The information contained in this chapter has been gathered from a variety of sources including industry interviews, primary data from the Australian Bureau of Statistics and secondary data from the Australian Bureau of Agricultural and Resource Economics.

Broad economic value of the industry: As outlined by the Bureau of Transport (2000) the effects of any economic activity are likely to reach beyond the initial round of output, income and employment generated by the activity.

For example, aquarium fish breeders can purchase inputs (e.g., equipment, fish feed) from domestic suppliers. The production of these inputs generates additional output, income and employment in the Australian economy.

The suppliers in turn purchase some goods and services from other Australian based firms. There are then further rounds of local re-spending as part of the chain of production.

Similarly, households that receive income from employment in the aquarium industry spend some of their income on local goods and services. These purchases result in additional jobs. Some of the household income from these additional jobs is in turn spent on local goods and services, thereby creating further jobs and income for local households. There are then further rounds of income generation as part of the chain of household expenditure.

As a result of these successive rounds of re-spending, the overall impact on the economy exceeds the initial round of output, income and employment generated by the industry.

The industry: The aquarium industry in Australia is a relatively small but growing sector of the economy. It comprises the retail sector (i.e., aquarium specific and broader pet stores) as well as the wholesale sector, which includes breeders, traders, importers, exporters as well as importers of aquarium-related products. There are also a number of associated sectors including the pet food sector, importers of aquarium products, importers of glass, cabinetmakers, nurseries (ponds) and small hobby breeders.

Trade and production: The Department of Agriculture Fisheries and Forestry recently valued the ornamental aquarium fish trade in Australia at approximately $350 million per annum (DAFF 2005). This figure included the input of commercial fish breeding facilities, wholesale traders, retail outlets and the hobby industry.

The 2001-02 value of aquarium and ornamental fish production levels in Australia was estimated by the Bureau of Agricultural and Research Economics as approximately $905,600 in 2001-02 (ABARE, 2003) This represents the value of production in Western Australia, Queensland, Victoria and New South Wales and includes both native and introduced fish species.

Breeding: There are several major breeders in Australia who service the domestic and international demand for Australian and non-native aquarium fish species.

Amongst the aquarium fish bred in Australia is a subset of exotic ornamental species, which include: angelfish, catfish, goldfish, koi carp, guppies, platys, mollies, rams, siamese fighting fish, swordtails, walking fish, red tiger oscars, gouramis and red rainbow fish.

Australian aquarium fish breeders have also taken an interest in producing native tropical species including: smelt, galaxiids, catfish, rainbowfish, hardyheads, perches, gudgeons and gobies

Juvenile food fish are also bred for the aquarium market. Species bred in Australia for juvenile food fish are barramundi, cod, and snapper.

The four states in which most of Australia's aquarium-based aquaculture occurs are, in descending order of value, Victoria, Queensland, New South Wales and Western Australia.

In 2001-02, the Victorian aquaculture industry produced approximately 3.9 million aquarium fish valued at $3 million. The sites of production are dams, ponds, flow through systems and recirculation units (NRE 2001).

In Queensland, the majority of aquarium fish are produced in re-circulated systems and ponds. In 2001-02 1.7 million exotic species valued at $741,000, and 342,000 native species valued at $121,000 were produced. Additionally, 1500 saratoga valued at $43,000 were grown (DPI 2002).

In the same time period the New South Wales industry produced 544,000 aquarium and ornamental fish valued at $338,000 (NSW Fisheries 2003), and Western Australia produced 288,000 aquarium fish in 2000-01 (Department of Fisheries 2002).

The relative importance of the various species is indicated in Table 7.1 (below) in terms of both their economic value to the industry and the estimated volume of fish sold. This assessment was made on the basis of discussions with J. Patrick of Bay Fish Wholesale Aquarium Fish Supplies, Narangba, Queensland.

Wholesale and retail turnover: According to industry estimates (Patrick 2001) the wholesale market was valued at $25 million per annum in 2001. Of this market 40% of stock is imported. A survey undertaken by the Pet Industry Joint Advisory Council (PIJAC) in 1999 provides the closest indication of the true value of the industry to Australia. For this report we have updated the information utilising a more current understanding of the retail sector.

A recent review of the retail market by analysis of the Yellow Pages listings of pet shops and aquariums recorded 1025 aquariums and pet shops in operation in Australia. Analysis of these data, utilising industry information gathered in 1999, indicates that there are approximately 6,150 staff employed in the aquarium retail sector and that the annual turnover is approximately $970 million (Table 7.2).

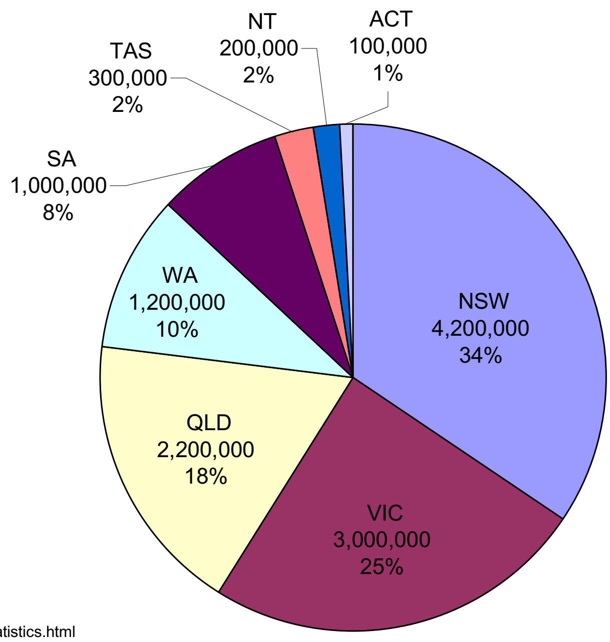
Consumer expenditure and pet fish population: Consumer expenditure on purchasing fish and the various goods and services relating to pet fish are between $75 and 90 million per annum (PIJAC communication).

BIS Shrapnel has estimated that the total pet fish population in Australia is approximately 12 million. Figure 7.1 indicates the distribution of ownership of pet fish across Australia.

Table 7.1: Relative importance of the aquarium fish species.

Common name	Scientific name	Relative importance	Volume of fish sold[1]
Family Cichlidae			
Hybrid cichlid	*Labeotropheus/Pseudotropheus*	(unknown)	Low
Jewel cichlid	*Hemichromis bimaculatus*	Medium	Low
Victoria Burton's haplochromis	*Haplochromis burtoni*	Low	(unknown)
Black mangrove cichlid	*Tilapia mariae*	(n/a)	Low
Redbelly tilapia	*Tilapia zillii*	(n/a)	(unknown)
Mozambique tilapia	*Oreochromis mossambicus*	(n/a)	(unknown)
Oscar	*Astronotus ocellatus*	High	Medium
Three-spot cichlid	*Cichlasoma trimaculatum*	Medium	Low
Jack Dempsey	*Cichlasoma octofasciatum*	Medium	Low
Red devil	*Amphilophus labiatus*	Medium	Low
Midas cichlid	*Amphilophus citrinellus*	Medium	Low
Convict cichlid	*Archocentrus nigrofasciatus*	Medium	Low
Blue acara	*Aequidens pulcher*	Medium	Low
Family Poeciliidae			
Green swordtail	*Xiphophorus hellerii*	High	High
Platy	*Xiphophorus maculatus*	High	High
Sailfin molly	*Poecilia latipinna*	High	High
Guppy	*Poecilia reticulate*	High	High
Caudo	*Phalloceros caudimaculatus*	Low	(unknown)
Family Osphronemidae			
Three-spot gourami	*Trichogaster trichopterus*	High	Medium
Family Cobitidae			
Oriental weatherloach	*Misgurnus anguillicaudatus*	(n/a)	(unknown)
Family Cyprinidae			
Goldfish	*Carassius auratus*	High	Very high
Rosy barb	*Puntius conchonius*	High	Medium
White cloud mountain minnow	*Tanichthys albonubes*	High	High

[1]Low = 10,000+; Medium = 10,000-100,000; High = 500.000-1,000,000; Very high >1,000,000



Source: http://www.petnet.com.au/statistics.html

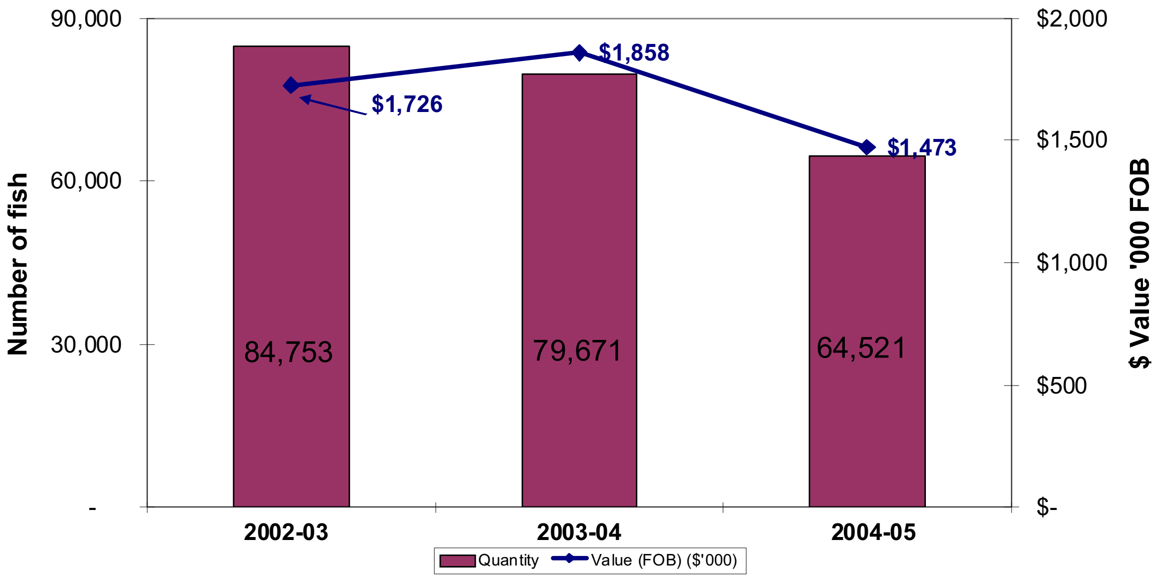
Figure 7.1: Distribution of pet fish ownership in Australia 2002.

Table 7.2: Aquarium retail sector 1999 and 2006.

	1999						2006					
	(A) No of stores	(B) % of total stores	(C) Annual Turnover	(D) Turnover per store	(E) Staff	(F) Staff per store	(G) No of stores	(H) % of total stores	(I) Turnover per store	(J) Annual Turnover	(K) Staff per store	(L) Staff
	*	= (A) / 793	*	= (C) / (A)	*	= (E) / (A)	**	= (G) / 1025	= (D)	=(I) x (G)	= (F)	=(G) x (K)
NSW/ACT	249	31%	$24m	$ 960,000	1570	6	337	33%	$ 960,000	$323m	6	2022
VIC	154	19%	$15m	$ 970,000	970	6	228	22%	$ 970,000	$221m	6	1368
QLD	182	23%	$17m	$ 930,000	1150	6	253	25%	$ 930,000	$235m	6	1518
SA	100	13%	$9m	$ 900,000	640	6	79	8%	$ 900,000	$71m	6	474
WA/NT	90	11%	$8m	$ 890,000	560	6	106	10%	$ 890,000	$94m	6	636
Tas	18	2%	$2m	$1,110,000	110	6	22	2%	$1,110,000	$24m	6	132
Total	**793**		**$75m**		**5000**		**1025**			**$970m**		**6150**

Source
* J Patrick (1999) *The Economic Impact of the Australian Aquarium Industry*
** Yellow Pages (2006) *www.yellowpages.com*

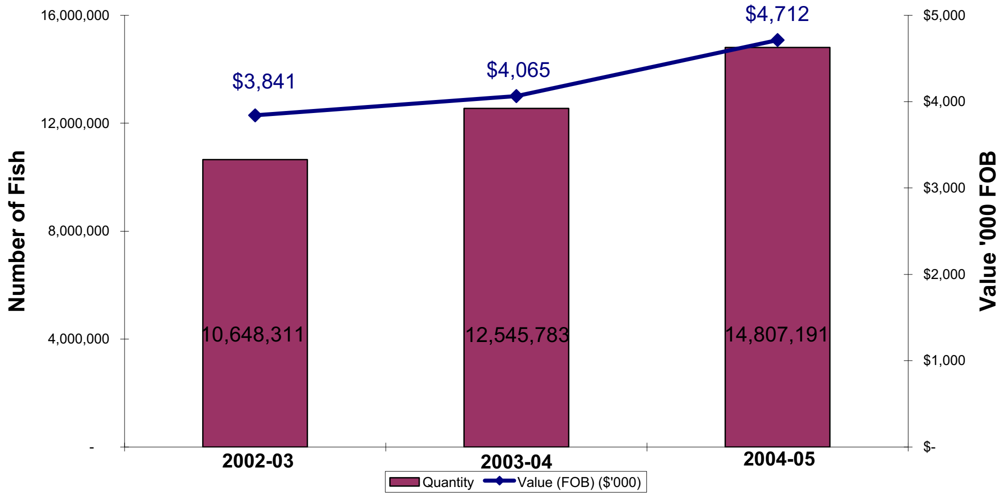
Trade: Over the past decade, the percentage of ornamental fish production (most of which are Australian natives) exported from Australia has undergone a significant decline. In 1995-96, 18.3 per cent of total production was exported overseas, whereas in 2000-01 the figure had dropped to 1.6 percent (ABS 2002;DPI 2002) Further, ABS data indicate that the value and quantity of aquarium and ornamental species exported from Australia have also declined in recent years Figure 7.2. In the 2004-05 financial year 64,500 fish (21,000 Australian species, 1,000 live syngathids and 42,000 non-Australian species) were exported at a value of $1.5 million. The main export markets were USA and Japan.



Source: Australian Bureau of Statistics (2005)

Figure 7.2: Quantity and value of aquarium and ornamental species exported from Australia 2002-03 to 2004-05.

This is in contrast to the value and quantity of imported species which were increasing over the same period (Figure 7.3). In 2004-05, 14.8 million fish were imported into Australia at a value of $4.7 million. These imports were predominantly from Indonesia and Singapore. This compares with $1.3 million for 9.7 million fish in the 1979-80 year (McKay 1984). Thus the number of ornamental fish imported has increased by 52% over the past 25 years and their value has increased by over 250%.



Source: Australian Bureau of Statistics (2005)

Figure 7.3: Quantity and value of aquarium and ornamental species imported to Australia 2002-03 to 2004-05.

Illegal trade: There is some evidence that there is a growing level of illegal trade of imported species in Australia. AQIS (1999) has estimated that the illegal import of species accounts for between 5-10% of the fish imported into Australia. This is supported by industry sources which suggested that the black market trade in illegally imported fish could be valued at up to $10 million per annum.

There are also species that cannot be imported legally, but are now present in Australia and, once here, can be freely traded. These species contribute to the value of the industry and its value would be affected if trade in these exotic species were to be restricted.

7.2 Australian studies of economic and social impacts

There are few Australian studies of economic and social impacts of introduced pest species, and no studies of the impacts of aquarium or ornamental fish establishing wild populations. Various technical papers consider the ecological impacts of introduced species and speculate as to the possible wider impacts. The absence of studies means that investigation of impacts and analytical approaches for assessing economic and social impacts are substantially unencumbered by the outcomes of other research.

Impact assessments of invasive species have been reviewed by Agtrans Research (Agtrans, 2005) for the Department of Environment and Water Resources. McLeod (2004) assessed the impacts of a range of invasive species in a 'Triple Bottom Line' framework (i.e., environmental, economic and sociological considerations are all considered). Substantial reliance was placed by Agtrans on the research of McLeod.

The invasive species considered by McLeod and the triple bottom line impacts are shown in Table 7.3. The results indicate a cost of in excess of $720 million. The only aquatic species included in the analysis is carp at a cost of around $16 million. A more detailed breakdown of carp costs is shown in Table 7.4. The main cost item is the environmental impact assessed to be $11.8 million.

Table 7.3: Triple bottom-line impacts of invasive species (Table is from the executive summary in McLeod 2004).

		Triple Bottom Line Impact					
	Total	Economic		Environmental		Social	
	$m	Impact	$m	Impact	$m	Impact	$m
Fox	227.5	◆	37.5	◆	190.0	♦	nq
Feral Cats	146.0	♦	2.0	◆	144.0	♦	nq
Rabbit	113.1	◆	113.1	♦	nq	♦	nq
Feral Pigs	106.5	◆	106.5	♦	nq	♦	nq
Dogs	66.3	◆	66.3	♦	nq	♦	nq
Mouse	35.6	◆	35.6	♦	nq	◆	nq
Carp	15.8	♦	4.0	◆	11.8	♦	nq
Feral Goats	7.7	♦	7.7	♦	nq	♦	nq
Cane Toads	0.5	♦	0.5	♦	nq	♦	nq
Wild Horses	0.5	♦	0.5	♦	nq	♦	nq
Camels	0.2	♦	0.2	♦	nq	♦	nq
Total	**719.7**		**373.9**		**345.8**		

nq = not quantified ◆ = bigger impact ♦ = smaller impact

The $11.8 million annual environmental cost was derived by aggregating an estimate of the cost of carp-related sedimentation and heightened water turbidity with a decline in recreational fisher value due to lower water quality and stocks of native fish. Other costs included are the direct costs of carp management and research. Carp-related turbidity and sedimentation costs were determined arbitrarily by assuming that 10 per cent of estimated annual costs of $24 million and $4 million respectively were attributable to carp (McLeod, 2004; p. 31). Justification for the assumption of 10 per cent is not provided and it appears to be based largely on conjecture.

The additional $9 million also appears to be similarly conjectural in its origin. McLeod (p.32) states, based on a survey of fishing in NSW,

"Given that somewhere in the order of 25% of fishers surveyed utilised inland waters, and many of the 5 million fishers in Australia would be irregular, it is estimated that there are around 0.6 million Australians who have regular contact with inland waters where carp could possibly be a problem. Aggregating the 'willingness to pay' for improved fishing quality of $50 per household over 0.6 million fishers, the aggregate cost of decreased fishing quality is estimated to be $30 million per year. This cost is derived on the basis, that in the absence of carp, fishers would have satisfactory water quality and greater abundance of native fish. If carp were contributing to a 30% decline in prized fish species, then a social cost of $9 million per year could be attributed to the impact of carp on recreational fisheries."

Table 7.4: Annual cost impact of carp**.**

Cost Component	Control $A million	Loss $A million	Total $A million
Management of carp [b]	2.00	-	2.00
Research Cost [c]	2.00	-	2.00
Environmental Impact [a]	-	11.80	11.80
TOTAL COST	4.00	11.80	15.80

(a): Annual cost to community estimated in this assessment
(b): Control costs for carp taken from Bomford and Hart (2002) and $1 million per year from the Tasmanian government for Crescent Lake
(c): Public sector research costs for carp taken from Bomford and Hart (2002) and new projects.

Source: McLeod (2004; p. 31)

Agtrans (2005) surveys the impacts of various invasive species. Impacts are classified as economic, environmental or social. Economic cost impacts of aquatic vertebrates, specifically carp, are identified as:

- Control
- Research
- Commercial and recreational fishing
- Water quality
- Tourism
- Decline in native fish species
- Agricultural – damage to irrigation channels (Agtrans, 2005; p.15).

The gross value of the carp industry in 2002 was specified as $1.7 million.

Agtrans (p. 16) comment on the additional estimates compiled by McLeod with reference to an estimate of carp costs deriving from the Gippsland Lakes and

Catchment Action Group of $35 million per year or $175 million over five years. McLeod (p. 32) noted this estimate but pointed out that, "the method for estimating these losses was not explained".

Although the focus of estimates is carp, as explained by Agtrans (p. 16),

"In addition to carp, there are a number of other introduced freshwater aquatic vertebrate species that have become invasive and that are having a negative impact on native fish and other aquatic species. Examples of these introduced species include:

- Eastern gambusia/mosquitofish (*Gambusia holbrooki*)
- Redfin perch (*Perca fluviatilis*)
- Rainbow trout (*Oncorhynchus mykiss*)
- Brown trout (*Salmo trutta*)
- Tench (*Tinca tinca*)
- Green swordtail (*Xiphophorus hellerii*)
- Mozambique tilapia (*Oreochromis mossambicus*)
- Oriental weatherloach (*Misgurnus anguillicaudatus*).

These species potentially have a negative economic impact in terms of reducing stocks of natural fish available for recreational fishing, and through general irrigation and agricultural impacts due to a reduction in water quality. However, there were no estimates identified of the economic impacts of introduced freshwater aquatic vertebrates other than for carp.

It should be noted that introduced fish species that are pests such as rainbow trout and brown trout, are also valued by recreational fisherman and provide some economic value through this industry (Agtrans 2005; pp. 15-16)".

There is some overlap between the environmental impacts and economic impacts identified by Agtrans (2005; p. 25):

"*Carp impact on commercial and recreational fishing, water quality, tourism, and on native fish species. Carp decrease water quality by contributing to increased nutrients, algae and suspended-sediment concentrations (Bomford & Hart 2002). This has a detrimental impact on aquatic plants and invertebrates. There may be some competition between carp and native fish for food and habitat, and carp may make aquatic habitats less suitable for other fish (Bomford & Hart 2002). Carp may have* *contributed to the decline of several threatened species including dwarf galaxias, trout, cod, Yarra pygmy perch and variegated pygmy perch (Bomford & Hart 2002)."*

The cost of the environmental impacts refers to the work of McLeod. In addition, Agtrans outline impacts attributable to other species:

Other introduced fish also have a negative impact on the environment. These include:

- Eastern gambusia/mosquitofish (*Gambusia holbrooki*) attack native fish, aggressively compete for food and prey on native fish and frog larvae. Reductions in native fish populations have been observed in most places where mosquitofish have been introduced (Arthington & Lloyd 1989; Bomford and Hart 2002).

- Redfin perch (*Perca fluviatilis*) are predators of native fish species (SoE SA 2003).

- Rainbow trout (*Oncorhynchus mykiss*) feed on a wide range of aquatic insects, crustaceans, molluscs, terrestrial insects and native fishes (SoE SA 2003).

- Brown trout (*Salmo trutta*) are aggressive predators of native fish, tadpoles and invertebrates (SoE SA 2003).

- Tilapia prey on native fish species and compete with them for food and habitat. They also remove plants. Tilapia pose a major threat to native fish species in Australia but are still in the early stages of establishing (Bomford & Hart 2002). However the tilapia is now considered well established in Queensland and it has already spread to the Burdekin Basin (A. Arthington, pers. comm.).

- Green swordtail (*Xiphophorus hellerii*) is an omnivorous feeder and there has been found to be a negative trend in the relationship between the abundance of *X. hellerii* and seven native species (Kailola 2000).

In an unpublished report to DEW, Kailola (2000) found that impacts on native fishes have been recorded by mosquitofish, swordtails, redfin perch, brown trout, rainbow trout, European carp, goldfish and possibly Oriental weatherloach. There are an additional fourteen established non-native fish species in Australia, and the effects of these species are unknown. Kailola (2000) found that the impact of non-native freshwater fishes on ecosystem functioning is still largely unknown, however there is circumstantial evidence of some impacts, as identified in the list above." (Agtrans 2005).

With regard to social impacts, Agtrans (2005) state:

"Water quality decline and reduction in native fish species leads to social impacts through reduced recreational fishing opportunities, limits on other water recreational activities, and tourism."

The pest status of several aquatic species are summarised in Table 7.5, and abundance and distribution of species relevant to this study are described in Table 7.6. Each of the species with a pest status of "serious" would be ideal candidates for a comprehensive, coherent and consistent study of economic, environmental and social impacts.

Table 7.5: Pest status of various aquatic species**.**

	Pest status		
	Serious	**Moderate**	**Minor or non-pest**
Freshwater Fish	European carp mosquitofish Mozambique tilapia	weatherloach tench redfin perch rainbow trout	brown trout goldfish guppy

Source: Agtrans (2005; p. 39).

Agtrans (2005; p.126) conclude that:

"Invasive species are costing Australia many billions of dollars annually mainly in costs of control and value of production foregone. Estimates of the different costs are incomplete and those that have been made need refinement and further justification if they are to be used to prioritise and stimulate further action on invasive species. The estimates made largely exclude the values of environmental or social costs of invasive species.

There is no commonly accepted method of valuing environmental impacts in dollar terms for purposes of priority setting among alternative activities and for integration with activities that lessen industry impacts. Willingness to pay methods of valuation have improved recently but are still used only sparingly by planners and policy makers. An additional issue is the adequacy of knowledge of the contribution of the invasive to any impact on native species or the wider ecosystem.

There are few studies that have identified in specific or quantitative terms the health, safety and quality of life/choice impacts of invasive species. A review could be undertaken of the seriousness of these impacts, particularly those involving human health and safety.

The benefits from invasive species need to be accounted for in more detail in the measurement of their costs so that a net cost to Australia can be estimated."

Table 7.6: Abundance and distribution of invasive aquatic vertebrate species.

Species	Origin, abundance and distribution
Carp	▪ Released on a number of occasions in 1800s and 1900s but not widespread until released in Murray River near Mildura in 1964 (McLeod 2004). ▪ Spread of carp through Murray Darling Basin coincided with widespread flooding in the early 1970s (McLeod 2004). ▪ Carp also were introduced to new localities – possibly through use as bait (McLeod 2004). ▪ Introduced carp are now the most abundant large freshwater fish in the Murray Darling Basin and are the dominant species in many fish communities in south-eastern Australia (McLeod 2004). ▪ Carp commonly found are from 50g to 5kg in weight and can tolerate a range of water temperatures, salinity levels and polluted water (Bomford & Hart 2002). ▪ A survey in 2003 found inland rivers had higher carp densities than coastal rivers. They were found in all inland sites surveyed below an altitude of 500 m above sea level (Bomford & Hart 2002). ▪ Carp are still expanding their range (SoE Qld 2003). ▪ Carp have broad environmental tolerances, thrive in disturbed habitats, can migrate at any time of year, move up to 230 km and are long living (PAC CRC 2004e).
Eastern Gambusia/ Mosquitofish (Gambusia holbrooki)	▪ Introduced in the 1920s for mosquito control – relatively ineffective for this purpose and now a significant pest in freshwater rivers and streams (SoE SA 2003).

Source: Reproduced from Agtrans (2005; pp. 45-46).

7.3 Modelling economic impacts and social impacts

Tensions and conflicts are commonplace when environmental issues are introduced into decision-making processes. A sense of entitlement based on a mis-apprehension of the nature and extent of property rights frequently colours the decision making process and deprives it of the required objectivity. Despite a history spanning more than 50 years, there remains a view that the inclusion of environmental impacts in economic analyses is an extension that is beyond the acceptable bounds of economics.

It is true that there is no single method that is suitable for all cases where values are assigned to environmental impacts. However, it is completely false to imply that there are no analytical tools that facilitate the assignment of acceptable dollar values to environmental impacts. Reputable and competent economic analyses have always attempted to account for externalities and many techniques have been developed and refined to facilitate the analysis. These techniques are not without inadequacies and are not beyond criticism; but they are no less adequate than many economic or other techniques that are relied upon for project analyses, or macroeconomic planning, or microeconomic planning (such as regulatory impact analyses).

The theory of externalities – positive or negative impacts of actions that extend beyond the direct market influence of the actions – is an integral part of economic theory and economic analysis of actions that impact upon the environment. Resistance to the application of a rigorous analytical framework to the evaluation of impacts owes more to the desire to protect sectional interests than it does to the adequacy or otherwise of the techniques used to assess the impacts. For example, the contingent valuation study used in the Exxon Valdez case was dissected and criticised to discredit this study in an attempt to reduce the large damages award. Where criticisms are directed at techniques or analytical frameworks it is important to consider who is making the criticisms, why they are making the criticisms, and what options are posited to overcome the inadequacies that are the basis of the criticisms.

The techniques discussed in this chapter are not designed to provide a means for decision makers to abdicate responsibility for making decisions to a number or a ratio. They are methods and techniques that are intended to assist the decision-making process through facilitating an objective quantitative and qualitative analysis of issues that results in balancing outcomes and to allow a decision-maker to arrive at a balanced decision.

7.4 Economic assessment methods

Various methodological frameworks can be used to undertake evaluations of economic, environmental and social impacts. The most common of these methods are cost-benefit analysis (now more commonly referred to as benefit-cost analysis (BCA)) and cost-effectiveness analysis (CEA). Other approaches include risk-benefit analysis (RBA), cost-utility analysis (CUA), multi-criteria analysis (MCA), decision analysis (DA), the Delphi Method (DM), and choice modelling (CM). Not all methods are mutually exclusive and elements of different methods may be combined to provide a comprehensive assessment. Further, not all techniques require the assignment of monetary values to impacts; rather they require that the analysis be explicit as to what impacts are monetised, what impacts are not, and the balance that is struck between the impacts that are quantitatively assessed and those that are qualitatively assessed.

The following discussion outlines the methods and where appropriate introduces impacts that might arise from aquarium species establishing wild populations.

Benefit-cost analysis (BCA): BCA is concerned with the analysis of a project or action from the perspective of society rather than an individual, firm or investor. This distinguishes it from a financial evaluation which considers only the financial costs and benefits relevant to the individual, firm or investor. That is, the boundaries of the analysis go beyond immediate market impacts to encompass incidental or external impacts.

As explained by Perkins (1994):

"An economic analysis, also called a cost benefit analysis, is an extension of a financial analysis. An economic analysis is employed mainly by governments and international agencies to determine whether or not particular projects or policies will improve a community's welfare and should therefore be supported."

For example, the information outlined above on the value of ornamental or aquarium fish industry provides very little insight as to the economic value of the industry. These values are gross values and should not be confused with the economic value which is a different concept and accounts for the fact that one area of economic activity attracts resources away from other areas of economic activity, and there are potential external impacts that might not be reflected in the market activities.

In a detailed study entitled Harmful Non-Indigenous Species in the United States, the US Congress' Office of Technology Assessment (OTA) investigated a wide range of introduced species in the United States. As outlined by the Director of OTA (1993) in the foreword:

"Non-indigenous species (NIS) – those species found beyond their natural ranges— are part and parcel of the U.S. landscape. Many are highly beneficial. Almost all U.S. crops and domesticated animals, many sport fish and aquiculture species, numerous horticultural plants, and most biological control organisms have origins outside the country. A large number of NIS, however, cause significant economic, environmental, and health damage. These harmful species are the focus of this study."

The issues and extent of the analysis that can be encompassed within a benefit-cost analysis framework are clearly illustrated in Figure 7.4. Although Australia is to some extent protected from invasive species by sea borders, in contrast to the United States, which has land borders with both Canada and Mexico, it is evident that many of the issues identified by OTA are relevant to Australian management of NIS, including harmful NIS.

Box 4-D-Outline of Steps for Benefit/Cost Analysis of Non-Indigenous Species

1. Effect estimation
   - A. Identify relevant input and output categories
     - 1. Inputs-(e.g., wetland invasion by non-indigenous melaleuca)
     - 2. Outputs-(e.g., tourism; honey production)
   - 6. Define units of measurement for input and output categories
     - 1. Inputs-(e.g., acres invaded)
     - 2. Outputs-(e.g., tourist expenditures; quantity of honey sold)
   - C. Establish a base of values for input and output categories without the introduction of the NIS
   - D. Identify production process relating to introduction of the NIS to a series of outputs, expressed probabilistically
     - 1. Expected units of invasion-(e.g., acres of distinct environs where NIS would be established and distributed)
   - E. Quantify expected magnitude of each output for the relevant magnitudes of each input category
   - F. Estimate changes in input and output categories for with introduction versus without introduction scenarios

ii. Valuation of direct effects
   - A. Market goods
     - 1. Marginal changes in production
       - a. Market price x change in output quantity
     - 2. Non-marginal change in product in product
       - a. Identify market price changes
       - b. Measure consumer and producer surplus
   - B. Non-market goods
     - 1. Contingent valuation

III. Calculate indirect effects
   - A. Multiplier income and employment effects
     - 1. Opportunity costs
     - 2. Unemployed resources
   - B. Related goods
     - 1. Changes in production
     - 2. Changes in market price
     - 3. Calculate consumer and producer surplus

IV. Calculate annual benefits and costs

V. Accounting for time
   - A. Select appropriate discount rate
     - 1. Use real (deflated) rate (e.g., riskless rate; Water Resources Council rate)
   - B. Convert annual benefits and costs to real terms (e.g., using CPI, GNP Deflator)
   - C. Calculate present value
     - 1. Present value of benefits = $\sum_{n=0}^{N} \frac{B_n}{(1+r)^n}$
     - 2. Present value of costs = $\sum_{n=0}^{N} \frac{C_n}{(1+r)^n}$

n. number of the year in time series, N = last year of time series, r = discount rate, B = benefits, C = costs

SOURCE: M. Cochran, "Non-Indigenous Species in the United States: Economic Consequences," contractor report prepared for the Office of Technology Assessment, March 1992.

Figure 7.4: Benefit-cost analytical framework inputs and outputs Source: U.S. Congress, Office of Technology Assessment (1993; p. 128).

Cost-effectiveness analysis (CEA): CEA is a technique that is used to either determine the maximum benefits that can be obtained from a specified expenditure, or to determine the minimum expenditure required to achieve a specified outcome. For example, in the control of a pest species, CEA could be used to maximise the impact of control for a given expenditure; or it could be used to determine the minimum cost required to achieve a desired level of control.

CEA can be used where there are *per se* obligations that are accepted in respect of policies or programs. Article 8(h) of the Convention on Biodiversity requires Parties to:

"Prevent the introduction of, control or eradicate those alien species which threaten ecosystems, habitats, or species;" (Article 8 (h), Convention on Biodiversity, entered into force on 29 December 1993; ratified by Australia, 18 June 1993).

Implementation of this article requires identification of alien species, specification of threats to ecosystems, habitats, or species, and prevention, control or eradication, of the alien species. Acceptance of the general obligation of the Article implies acceptance of the required consequential actions suggesting that CEA would be a suitable method for maximising benefits or minimising costs associated with implementation of the obligations.

Reflecting the potential usefulness of the CEA framework, the recently released draft management plan for ornamental fish (DAFF 2005) observes that:

"Of the 34 alien fish species that have established feral populations in Australian waters, 22 are thought to have come into the country via the ornamental fish trade (Lintermans 2004). It is commonly accepted in invasive species management theory that eradication of species once they are established is difficult, if not impossible, and that the most (cost) effective management is achieved through the prevention and management of introduction and spread."

Risk-benefit analysis (RBA): RBA is a technique that explicitly recognises within a benefit-cost framework that many outcomes are characterised by risk; that is, the risk of various outcomes can be quantified (assigned probabilities) and expected values (impact of the outcome multiplied by the probability of its occurrence) rather than market values included in the analysis. This contrasts with uncertainty where probabilities cannot be quantified and assigned. In this case, other techniques are required.

The potential importance of RBA for application to ornamental fish is reflected in the comments of Koehn (2004).

"Although our understanding of the impacts of alien fish is poor, and there is a lack of coordination, a review of the literature shows there is a range of information available that could form the basis of improved management of alien freshwater fish species in Australia. This information is of three types: (1) general strategic documents; (2) area based assessments; and (3) reviews of individual species. However, a coordinated approach such as that outlined for marine pests (National Taskforce on the Prevention and Management of Marine Pest Incursions 1999) is needed.".

RBA can facilitate the inclusion in any analyses of various impacts that might be omitted. In addition, analysis of issues that depend on biological and ecological systems and influences requires the use of different methods from those that would be applied in other areas. For example, often emission of a pollutant from an industrial process is linearly related to output and pollution control options are clearly defined, enabling a reasonably direct assessment of abatement costs and abatement benefits. Clearly, there are issues related to the extent of pollution plumes, and the rate of dispersion and assimilation of plumes. The rate of generation of pollution and total amount of pollutant can be reasonably well defined.

By contrast, assessment of the impacts of invasive species is more complex and will depend on an array of factors and interactions. Eldredge (2000) citing the work of Ehrlich (1986) identifies:

".....eight ecological, genetic, and physiological characteristics that might lead to successful introduction:

1. Abundant in original range.
2. Polyphagous.
3. Short generation time.
4. High genetic variability.
5. Fertilised females able to colonise alone.
6. Larger than most relatives.
7. Closely associated with humans.
8. Able to function in a wide range of physical conditions."

Investigation of species' impacts needs to start with an evaluation of the species' population dynamics, which requires analysis of reproduction, survivability, spread and consequential impacts. Simberloff (1996) reflects on the fact that:

"Introduced species cause disasters that one would never have foreseen. It might not seem surprising that the spread of fire-adapted, exotic plants that burn easily has increased the frequency and severity of fires, to the detriment of property, human safety, and native plants and animals. But would one have guessed that, in 1936, the town of Bandon, Oregon would be destroyed and eleven citizens killed by a fire propagated by gorse, a highly flammable plant introduced, seventy years earlier, from Europe?"

Rather than the impacts not being foreseen, it is more likely that there was no attempt to investigate impacts or quantify the risk of various outcomes.

In similar vein, Simberloff (1996) continues:

"Costs of introduced pathogens and parasites to human health and the health of economically important species have never been comprehensively estimated, but must be enormous. A recent example is the Asian tiger mosquito, introduced to the U.S. from Japan in the mid 1980s and now spreading in many regions, breeding largely in water that collects in discarded tires. The species attacks more hosts than any other mosquito in the world, including many mammals, birds, and reptiles. It can thus vector disease organisms from one species to another, including into humans. Among these diseases are various forms of encephalitis, including the La Crosse variety, which infects chipmunks and squirrels. It can also transmit yellow fever and dengue fever."

The comments of Simberloff need to be balanced against the fact that many introduced species are benign. Ciruna et al. (2004) note that:

"......., the great majority of introduced species do not cause problems of any sort. Most ornamental plants do not establish themselves outside gardens, and most species of discarded or escaped pets cannot survive in the wild. Of the minority of introduced species that do live for long outside human-dominated habitats, many are not invasive."

Estimation of population dynamics is based on stochastic (probabilistic) models. Under well-specified conditions, these models describe how a population is expected to reproduce and spread. The results can then be extended to practical situations and incorporated into an economic analysis using the RBA method. This appears to be the purpose of bioeconomic modelling. Choquenot et al. (2004) explain the *process:*

"Although the capacity to formally analyse management options for invasive species is clearly of benefit to a range of policy makers, the emphasis that bioeconomic analysis places on the development of conceptual, analytical, and/or simulation models produces a range of collateral benefits. These include:

- *A structured analysis of the problem—model development requires a clear articulation of the impacts a pest species is thought to have, who the beneficiaries of control are, and what the consequences of not controlling the pest will be.*

- *A review of existing data and information—model development involves a formal analysis of critical information gaps that exist concerning the pest, its control, and its impacts. As such, bioeconomic analysis can be used to* *prioritise research questions and identify critical monitoring points in the management systems.*

- *A tool for integrating new information and data as they come to hand—the development of bioeconomic models provides a framework for integrating new information and data as it comes to hand. By ensuring that the best available information is always available to managers and policy makers, these models become the primary mechanism for ensuring best practice management and decision making. Models can also provide an "institutional memory" of why particular policy positions were adopted, or management decisions made."*

A detailed schematic of a bio-economic framework using stochastic dynamic programming (SDP) is shown in Figure 7.5. The complexity of feedback interactions between ecological, economic and objective function optimisation is clear.

Multi-criteria analysis (MCA): The objective of reducing impacts to monetary values is to enable comparisons and reconciliations based on a common metric. This is not always possible nor is it desirable to force outcomes where the establishment of a common measure is unachievable. MCA is used where various inputs and outputs cannot be reduced to a common metric and are incommensurable. In order to take explicit account of these impacts, some system of ranking needs to be devised in order to enable comparisons. The ranking method is determined based on importance weights. Assigning importance weights is a subjective exercise but cannot be avoided unless better information is available. It might be thought that given the subjectivity of the exercise the problem can be solved by omission; but omission assigns a weight of zero.

The advantage of MCA is that it forces an explicit balancing of incommensurable outcomes about which investigators can then debate.

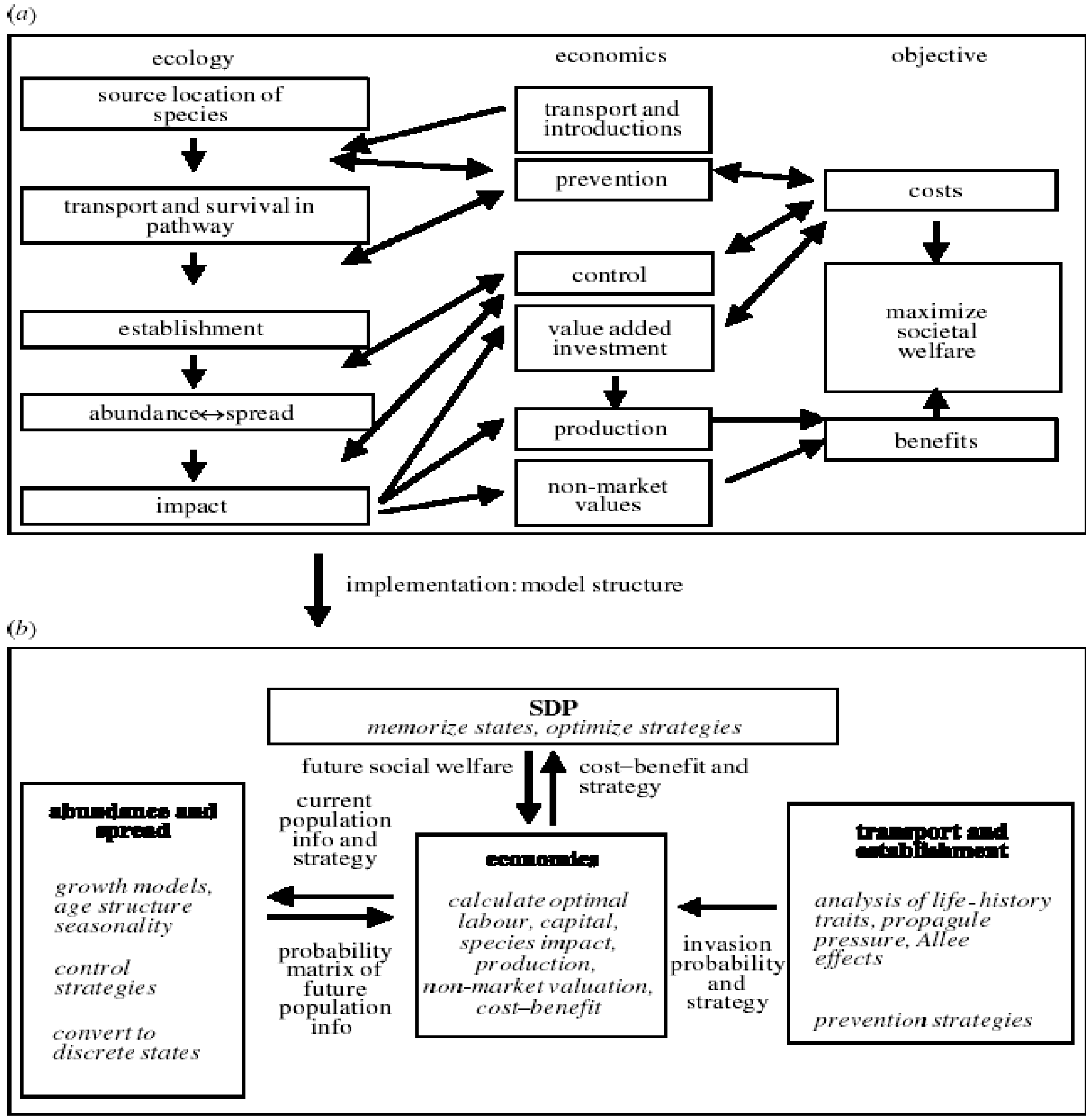


Figure 7.5: Bio-economic framework for invasions: (a) the conceptual approach to the ecological and economic components of a generalized invasion process. Both economic input and ecological states change over time and influence one another. Our goal is to determine the optimal set of strategies that maximize welfare, where welfare can be a function of both market and non-market values, (b) Implementation of the conceptual approach through an operational model structure. The boxes and bold text represent modules, within which details (italic text) may be hidden (encapsulated) and modified without affecting the entire model. Plain text represents the interfaces (information passed between modules). (Figure is from Leung et al. 2002).

Decision analysis (DA): The SDP bioeconomic framework can be interpreted as an extension of BCA or RBA, incorporating all of the elements of these methods and extending these to recognise that there are different stages of decision making whereby different states of nature (outcomes) can be characterised. This process is called decision analysis (DA) and it defines various strategies and actions along with associated outcomes. Where possible probabilities are assigned to the outcomes and expected values of costs and benefits from different strategies can be calculated and compared. The basis of DA is the construction of a decision tree similar to the framework used for stochastic dynamic programming applied to both bioeconomic models (discussed above) and stochastic resource models (see, for example, Conrad and Clark, 1987).

DA does not specify a rule for choosing between strategies; rather it is left to the decision maker to determine which strategy to pursue.

Delphi method (DM): The DM has typically been employed as an alternative to pure quantitative modelling and analysis. It relies upon group decision-making using a panel of analysts who are experts in the area to be investigated. The technique typically consists of four stages (Linstone and Turoff, 1975):

- establishing the components and parameters of the policy or project;
- formulation of views, including points of view on importance, desirability or feasibility of proposed actions;
- exploration of issues of significant disagreement; and
- final evaluation, including reasons for agreement and disagreement.

Within economics, citizens' jury and choice modelling can be seen as adaptations of the DM. The Citizens' jury method is explained by Robinson et. al. (n.d.; p.5):

"*Citizens' jury is a deliberative form of public participation. This approach is an effective way to involve citizens in developing a thoughtful, well-informed solution to a public problem or issue. The Citizens' jury is based on the model used in Western-style criminal court proceedings. The great advantage of the Citizens jury process is that it yields citizen input from a group that is both informed and representative of the public*."

Bennett (2005) summarises the elements of choice modelling as:

"Choice modelling (CM) is a 'stated preference' technique that can be used to estimate non-market environmental benefits and costs. It involves a sample of people, who are expected to experience the benefits/costs, being asked a series of questions about their preferences for alternative future resource management strategies. Each question, called a 'choice set', presents to respondents the outcome of usually three or four alternative strategies. The alternatives are described in terms of a common set of attributes. The alternatives are differentiated one from the other by the attributes taking on different levels. One of the alternatives – that relating to the 'business as usual' (BAU) option – is held constant and is included in all the choice sets."

7.5 Defining inputs and deriving values

Implementation of the methods, outlined previously and which incorporate monetary values, requires estimation techniques that allow the assignment of values to impacts. This section discusses various approaches that can be used. These approaches fall into three broad categories: market-based techniques, surrogate-market techniques, and survey-based techniques. Market-based techniques rely upon market transactions to identify and quantify values of environmental goods and services. Surrogate-market techniques depend on proxy values determined from disaggregation of the characteristics of trades within markets. Survey-based techniques attempt to determine values through constructing a theoretical market. Essentially, the surrogate-market and survey-based techniques recognise that many environmental goods and services are not, and cannot be, traded directly – markets are missing – and alternative approaches to valuing these goods and services are essential if they are to be properly accounted for in analyses. As with other areas, the delineation between techniques is not strict.

(A) Market-based techniques

Productivity changes: Ornamental fish establishing wild populations can impact on the productivity of other industries. For example, increased turbidity and sedimentation attributable to carp can promote growth of blue-green algae poisoning stock water supplies and reducing the productivity of farms. Stock could be poisoned or lose condition as a result of ingesting the affected water. This has a direct impact on the market value. Other impacts could include the impairment of productivity of existing fisheries. The market values derived from these productivity impacts can be used as measures of the costs of environmental impacts.

Opportunity cost: In order to preserve an environmental resource, expenditures are required and these expenditures have an opportunity cost. That is, income is foregone from other market-based uses of the resource. In this context, resources used in monitoring and controlling invasive species have an opportunity cost that can be assessed and included in an analysis.

Preventive expenditures: These are expenditures that are made in order to prevent or avert environmental damage. Expenditures on monitoring and control programs can be characterised as preventive expenditures on the basis that the purpose of the expenditures is to prevent environmental damage rather than to simply monitor the extent of environmental damage. The preventive expenditure can be construed as the minimum value of the environmental resource.

Replacement and repair costs: Some environmental impacts result in the complete destruction of an environmental resource or serious degradation of the resource over a long period of time. In the case of destruction, a measure of the value of the resource is the cost of replacing the services that have been eliminated. This does not necessarily involve restoration of an identical resource; merely the replacement of the destroyed resource with one that delivers an equivalent stream of goods or services. Repair or rehabilitation cost measures are derived based on the cost of rehabilitating the degraded resource to bring it back to a level of functionality existing prior to the degradation.

Shadow or compensation project approach: Shadow or compensation project valuations are based on estimates of the cost of a project that is provided as compensation for the degradation of an environmental resource. The compensation project can be seen as a special case of the replacement cost approach and involves two key assumptions (James, 1994):

- The value of the endangered environmental goods and services is marginally greater than the costs of the shadow project.

- The shadow project can adequately replicate the endangered environmental goods and services.

Relocation cost: This involves investigation of the costs of relocating activities affected by the degradation of environmental resources. For example, if environmental degradation undermines tourism operations but these can be relocated in another area through expenditures on suitable infrastructure elsewhere, these expenditures can be used to indicate the cost of the environmental degradation.

Surrogate-market techniques: Surrogate market techniques are used to estimate environmental values where there are no direct markets for the environmental good or service, but it is clear that they have a value based on expenditures incurred by individuals in taking advantage of the good or service. The techniques draw on and analyse information about jointly consumed products to estimate the economic value of the resource in its current state. A relationship between the resource availability and economic value is the end product of surrogate market techniques.

Hedonic pricing technique: Hedonic pricing defines goods and services based on their attributes or characteristics. The technique is used to assign environmental values through disaggregating attributes associated with a good or service, part of the bundle of attributes being environmental. For example, housing that is directly under a flight path would be expected to have a lower value than housing that is unaffected by aircraft noise. Similarly, property adjacent to an undisturbed physical environment would be expected to have a higher value than similar property located within sight and sound of a mining operation or landfill.

Application of the technique requires the following steps:

i. identify the market good or service (usually property) and the environmental good or service of concern;

ii. define a functional relationship between property price, and the property attributes that contribute to the property price, including the structural features of the property, any relevant neighbourhood characteristics, and the environmental attribute of concern;

iii. collect data that are used in the functional relationship, either for a large number of properties at one point in time, or for a smaller number of similar properties over a number of years; and

iv. estimate the functional relationship, using econometric techniques, and estimate the contribution of the environmental attributes to the property price. (Aquatech, 1996).

Travel cost method: In order to take advantage of environmental goods and services, individuals expend resources on accessing these goods and services. Both direct expenditures, fuel, wear and tear on vehicles, and indirect costs based on the value of time, are incurred. James (1994) outlines the procedure as follows:

i. Site visitors are surveyed to ascertain the frequency of visits from zones of origin. For example, if the recreation site was clear, series of concentric circles can be drawn spreading out from the site. Each band of territory would constitute a potential visitor origin zone. The visitation rate for each zone of origin is determined by dividing the number of visitors from each zone by the respective zonal population. Population figures for each zone must be obtained from independent sources of data.

ii. Travel costs to the site are determined for each zone. Travel costs should include all costs of reaching the site, including the cost of travel time.

iii. Visitation rates are regressed on travel costs across all zones to obtain the travel cost function. This function can be used to estimate visitation rates as a function of ‘price’ paid. Initially, the price paid by each zone will be the travel cost itself.

iv. The assumption is then made that travel costs act as a proxy for admission charges to the site. An admission charge can be added to the travel cost for each zone and, using the travel cost equation, it is possible to ‘predict’ the visitation rate for each zone.

v. For each simulated admission price, the predicted number of visits from each zone can be found by multiplying the population in each zone by the corresponding visitation rate. Total visits to the site, for the given admission price, can be determined by aggregating predicted visits across all zones. This gives one point on the implicit demand curve.

vi. By repeating steps (4) and (5) the demand curve for the site amenity can be constructed. The marginal willingness-to-pay (WTP) (admission price) is given on the vertical axis and the number of visits on the horizontal axis.

vii. Assuming a zero price if charged, total user benefits will consist of consumers' surplus under the demand curve. The final figure represents the total WTP for use of the site amenity. This value can be left as an annual benefit from the site, or it can be capitalised into a present value equivalent by dividing it by the appropriate discount rate."

(B) Survey-based and panel techniques

Contingent valuation method: The contingent valuation method (CVM) is a survey-based method that is used to assign values to environmental goods and services where no markets exist. CVM uses two related concepts – willingness-to-pay (WTP) and willingness-to-accept (WTA) – in order to assign values to environmental goods and services. WTP is used to determine the amount an individual would be willing to pay to prevent a clearly specified deterioration in an environmental good or service, and WTA is used to estimate the amount an individual would accept in compensation for agreeing to a clearly specified deterioration in an environmental good or service.

CVM has been subject to criticisms on the basis that because there are no market transactions ultimately resulting from the exercise, there is an incentive for individuals to exaggerate the amount they are willing to pay to preserve an environmental asset or the amount they are willing to accept for the loss of an environmental asset. This results in over-estimation of environmental values.

Mitchell and Carson (1989) provide a detailed account of CVM, and Diamond and Hausmann (1994) provide an extensive critique. As with any survey-based method, survey design is of critical importance as is recognition of potential problems. With CVM, the good or service must be familiar, the means of payment needs to be explained, and the valuation process has to be believable. Mitchell and Carson (1989) observe that the means of payment should be realistic and neutral.

Citizens' jury and choice modelling: Citizens' jury is explained by Robinson et. al. (n.d.):

"Citizens' jury is a deliberative form of public participation. This approach is an effective way to involve citizens in developing a thoughtful, well-informed solution to a public problem or issue. The Citizens' jury is based on the model used in Western-style criminal court proceedings. The great advantage of the Citizens jury process is that it yields citizen input from a group that is both informed and representative of the public."

Bennett (2005) summarises the elements of choice modelling as:

"Choice modelling (CM) is a 'stated preference' technique that can be used to estimate non-market environmental benefits and costs. It involves a sample of people, who are expected to experience the benefits/costs, being asked a series of questions about their preferences for alternative future resource management strategies. Each question, called a 'choice set', presents to respondents the outcome of usually three or four alternative strategies. The alternatives are described in terms of a common set of attributes. The alternatives are differentiated one from the other by the attributes taking on different levels. One of the alternatives – that relating to the 'business as usual' (BAU) option – is held constant and is included in all the choice sets."

"CM, as a stated preference technique, requires the collection of primary data. This in turn requires the use of a survey. The smallest CM exercise would normally require a sample size of around 1000 valid responses for it to provide sufficient statistical power. However, smaller samples are possible where respondents may be expected to answer a greater number (more than eight) of choice sets in each questionnaire. This is likely to occur when the issue of interest directly affects respondents (e.g., a local issue)".

7.6 Knowledge gaps and experimental designs to address knowledge gaps

As observed in the introduction, knowledge of social and economic impacts is substantially unencumbered by the results of previous research. Impact values appear to be based largely on conjecture and relate to carp only. Other species that have or could establish wild populations do not seem to have been subjected to any level of rigorous economic or social investigation or analysis in a coherent framework that takes advantage of detailed technical knowledge arising from scientific understanding of potential ecological impacts of ornamental fish species. Overall there is no consistency in either the formulation of the problem:

"Attempting an objective analysis and summary of the studies (of economics of biological invasions) that have been done is frustrating, as every study has used a different approach, making an accurate assessment of aggregate impacts impossible" (Wilgen et al. 2001).

Designing an experiment or experiments to address knowledge gaps needs to encompass both biology and economics with the starting point being an operational characterisation of the population dynamics and spread of the fish species selected for study. However, the objective of this section is not to present a definitive design but to outline an approach to addressing knowledge gaps.

The standard model of population dynamics relates the change in a population to the starting population:

$$\frac{dP}{dt} = kP \qquad (1)$$

where P is the initial population, t is time and k is a constant of proportionality.

Growth will be limited by the capacity of the receiving environment with population converging to a stable population based on this carrying capacity. Defining N as the stable or threshold population, as population approaches this value the growth rate will decline to zero:

$$\frac{dP}{dt} = kP(1 - \frac{P}{N}) \quad (2)$$

For P much smaller than N, 1-P/N is approximately one (P/N » 0) and for P=N, 1-P/N = 0. Equation (2) is called the logistic growth model where the term logistic has no particular meaning.

Characterisation of population growth models is not standardised and is apt to cause confusion. Leung et. al. (2002) formulate their model as:

$$\frac{dN}{dt} = rN(\frac{1-N}{\mathrm{K}}) + \varepsilon \quad (3)$$

where N is taken to be population, r is a growth rate, K is the limiting value, and e is uncertainty or a disturbance term. Apart from e, the other terms are not defined in Leung et al. (2002) and the model appears to be problematic. As N → K, dN/dt → 0 in the logistic model; however, in Leung et al.'s model, the growth rate, dN/dt → r(1-K), that is the exponential growth rate, r, from which is subtracted rK (in order to maintain a stable population over time at K, rK could be interpreted as replacement, but Leung et al.'s incomplete definition of the problem does not provide adequate guidance as to the analytical intention of the formulation). Choquenot et al. (2004) present another formulation of the logistic growth model:

$$r = r_m(1 - \frac{N_{t-T}}{\mathrm{K}}) \quad (4)$$

where r is the exponential growth rate, Nt is prevailing population abundance, rm is the maximum exponential rate of population growth, T is a time lag, and K is the limiting value of population at carrying capacity. Equation (4) is a consistent formulation in that as:

$$N_{t-T} \rightarrow K$$
$$r \rightarrow 0$$

That is, as the population approaches carrying capacity, the exponential growth rate tends to zero.

Both Leung et al. (2002) and Choquenot et al. (2004) extend their models to analyse economic impacts of invasive species management and control. Leung et al.'s analytical framework is illustrated in Figure 7.5. Welfare is defined in terms of society's profit function, which is not the traditional definition of welfare; and production is specified to be according to a Cobb-Douglas functional form. Choquenot et. al. extend their analytical framework to benefit-cost analysis which, more correctly as they define it, is cost-effectiveness analysis (benefit maximisation or cost minimisation).

Experimental design:

- Define the species to be investigated and the investigation area.
- Characterise the population dynamics of the population.
- Specify the area that will be impacted by the population. The area of impact is a key issue in that the wild population is of interest only to the extent of its spread which can be defined in terms of the area of impact; for example, kilometres of stream/river, hectares of marshland, degree of exclusion of existing species, etc.
- Identify and classify the impacts, with the starting point being the ecological impacts. An alternative would be the Convention on Biodiversity which imposes a per se obligation that can be taken as a starting point.
- Specify the objectives of the investigation - control of the spread of the species, eradication of the species.
- Characterise different levels of control and the ecological and economic impacts that are associated with each level.

Table 7.7 summarises various ecological impacts associated with invasive alien species that can be used as a starting point for identification of impacts that have potential social and economic consequences.

Table 7.7: Examples of the ecological impacts of invasive alien species (including both aquatic plants and fish) on inland water ecosystems.

Ecological Factors	Impacts
Change in Physical Habitat	Loss of native habitat.
Change in Hydrologic Regime	Alteration of surface water flow regime. Alteration of groundwater regime. Alteration of soil moisture regime. Alteration of evapotranspiration regime.
Change in Water Chemistry Regime	Alteration of dissolved oxygen concentration(s). Alteration of dissolved mineral concentrations. Alteration of dissolved organic matter. Alteration of turbidity.
Change in Connectivity	Alteration of lateral connectivity (e.g., river – floodplain connectivity), longitudinal connectivity (e.g., upstream – downstream connectivity), vertical connectivity (e.g., river - groundwater connection through the hyporrheic zone).
Biological Community Impacts	Loss of native species diversity. Alteration of native trophic structure and interactions. Alteration of native biomass.
Species Population Impacts	Loss of or decrease in native species populations through predation. Loss of or decrease in native species populations through competition for food, shelter, habitat and other important resources. Loss of or decrease in native species populations through pathogens/parasites carried by invasive alien species. Dispersal/relocation of native species populations through over-crowding and aggressive behaviour. Decrease in reproduction rate and fecundity of native species populations. Decrease in growth rates of native species populations. Alteration of behaviour in native species populations.
Genetic Impacts	Loss of genetic variability through hybridization. Loss of genetic variability through introgression/gene-swapping (i.e., erosion of the native species population's gene pool).

Source: Ciruna et al. 2004; pp.33-34.

Rather than the models of Leung et al. (2002) and Choquenot et al. (2004) it might be better to specify an alternative model following that devised by Perrings (n.d.) which is perhaps a better approach to analysis of management and control of invasive species in that it explicitly recognises the balance between invasive species and native species which can facilitate the balancing of damage costs against control costs. For example, the model of population dynamics is the starting point as defined above:

$$\frac{dP}{dt} = kP(1 - \frac{P}{N}) \qquad (5)$$

N is the limiting value of the population or carrying capacity of the environment which can be redefined in terms of the area, A, occupied by the invasive species. Assuming that A is directly proportional to P, the problem can be formulated in terms of the area:

$$\frac{dA}{dt} = cA(1 - \frac{A}{\mathrm{M}}) \qquad (6)$$

where A is as defined, c is the constant of proportionality, and M is the maximum area of invasion. As with population, as A→M, the increase in space occupied tends to zero. Further as the invasive species occupies more of the area it will:

- exclude existing species;
- impact on habitat;
- potentially change the balance between the decision to eradicate compared with control; and
- result in changing ecological, social and economic impacts.

Several values of A can be defined which will result in different responses, where A can be defined in terms of hectares, kilometres of stream, etc. Perrings (u.d) explains that the control of invasives includes a number of options: exclusion, eradication, containment (control), mitigation and adaptation. As A tends towards a particular value, less than M, the choice of management option can shift between, for example, eradication to containment. That is, the following scenarios can be specified:

Area occupied – A1; option – exclusion.

Area occupied – A2; option – eradication.

Area occupied – A3; option – containment (control).

Area occupied – A4; option – mitigation and adaptation.

For each A, there will be ecological impacts, which will ramify into social and economic impacts. The value of the growth rate function will change as A moves from A1 to A2, etc. Noting that c is the relative growth rate and solving the differential equation for the logistic growth equation:

$$A = \frac{M}{1 + \alpha e^{-ct}} \quad (7)$$

Equation (7) can be solved for each value of A to yield the relative growth rate. In turn, this provides information on the rate at which the space is being invaded which leads to specification of the ecological impacts and threshold levels where management options switch between exclusion to eradication to containment, etc. Under each option benefits and costs can be specified deriving from the identification of ecological impacts as outlined, for example, in Table 7.7. These can then be analysed within a cost effectiveness analysis framework or benefit-cost analysis framework. The issues and extent of the analysis that can be encompassed within a benefit-cost analysis framework are illustrated in Table 7.8.

Table 7.8: Benefit-cost analytical framework inputs and outputs.

I. Effect estimation	A. Identify relevant input and output categories: 1. Inputs (e.g., wetland invasion by non-indigenous species 2. Outputs (e.g., tourism, honey production) B. Define units of measurement for input and output categories: 1. Inputs (e.g., acres invaded) 2. Outputs (e.g., tourist expenditures, quantity of honey sold) C. Establish a base of values for input and output categories without the introduction of the NIS. D. Identify production process relating to introduction of the NIS to a series of outputs, expressed probabilistically: 1. Expected units of invasion (e.g., acres of distinct environs where NIS would be established and distributed). E. Quantify expected magnitude of each output for the relevant magnitudes of each input category. F. Estimate changes in input and output categories for 'with introduction' and 'without introduction' scenarios.
II. Valuation of direct effects	A. Market goods 1. Marginal changes in production a. Market price x change in output quantity 2. Non-marginal change in production a. Identify market price changes b. Measure consumer and producer surplus B. Non-market goods 1. Contingent valuation 2. Citizens' jury 3. Choice modeling
III. Calculate indirect effects	A. Multiplier income and employment effects 1. Opportunity costs 2. Unemployed resources B. Related goods 1. Changes in production 2. Changes in market price 3. Calculate consumer and producer surplus
IV. Calculate annual benefits and costs	(= outcome of steps outlined above)
V. Accounting for time	A. Select appropriate discount rate 1. use real (deflated) rate (e.g., risk-free rate) B. Convert annual benefits and costs to real terms C. Calculate present values 1. Present value of benefits = $\sum_{n=0}^{N} \frac{B_n}{(1+r)^n}$ 2. Present value of costs = $\sum_{n=0}^{N} \frac{C_n}{(1+r)^n}$ n = number of years in time series; N= last year of time series; r = discount rate; B_n = benefits; C_n = costs.

Source: Adapted from U.S. Congress, Office of Technology Assessment (1993).

8. Impacts from aquarium fish in relation to other stressors

8.1 Introduction

Any impacts of aquarium fish need to be considered alongside those created by other exotic fish including salmonids (trout), common carp, perch, and gambusia (mosquitofish). They also need to be placed in the context of impacts from other stressors such as altered flow regimes, the deterioration of water quality, the reduction in habitat for fish, and the effects of dams on fish migrations and hence recruitment.

The social and economic impacts of exotic fish species other than aquarium fish have already been discussed in chapter 7.2. It is clear from this discussion that the costs and values of these fish can be more easily appraised than those of aquarium fish, principally because a lot more is known about the uses, impacts and management of the non-aquarium fish species. The lack of information on aquarium fish impacts, and the fact that most aquarium fish are currently known from far fewer locations, severely limits any quantitative comparison.

This aside, it might be argued that the impact of aquarium fish as a whole on the native fish fauna will be much less than that of introduced fish such as the salmonids and perch, because the latter species are larger, are specialised piscivores, are more widely distributed and at present are more actively spread (e.g., through stocking). Because of these attributes they have arguably had a much greater and widespread impact on native fish than the aquarium fish species. However, the impact of gambusia (mosquitofish) on small native fish throughout the world indicates that piscivory is not a pre-requisite for impacts by exotic fish on native species. Similarly, the common carp is not a piscivore and yet under some circumstances it may generate major changes in environments, which then affect the native fauna. A number of aquarium fish in Australia have similar behavioural characteristics to gambusia and common carp and therefore have the potential to cause impacts related to those caused by these pest fish species. Therefore, aquarium fish may too contain the potential for measurable, widespread impacts.

Despite the lack of evidence that aquarium fish are currently impacting on the native fauna, there is enough now known about the behaviour of some aquarium fish species to create real concern over their potential to cause impacts, especially if they occur or are spread more widely. The real comparison between these two groups of fish should be between their overall potential impact some time in the future assuming that the more dangerous species will spread further. At present, it can be argued that aquarium fish have less of an impact than other exotic fish species because they are not as widely spread and their impacts are less well known. However, should they spread more widely over the next century and impacts on native fish be shown to occur, then their impact may well grow to be of a similar order of magnitude to that of other exotic fish. The difference will be that they may occur in the more northern and hence warmer waters of Australia than in the southern regions.

A comparison between the overall impacts of aquarium fish and other stressors of freshwater ecosystems is more difficult primarily because of a lack of detailed information on how these other stressors affect native fish. A related problem is the lack of information on the distribution of both stressors and fish. To overcome these difficulties we carried out a qualitative benchmarking exercise. This assessed the impacts of selected stressors along a number of gradients including spatial scale, impact type and severity and management costs. This is not an exhaustive or comprehensive approach as required by the economic modelling recommended in chapter 7, but it provides a first attempt to place the potential impacts of aquarium fish 'in context'.

8.2 Methods

We selected 5 major environmental stressor categories for benchmarking against the impacts of established ornamental fish. These were as follows:

- Altered flow regimes.
- Degraded water quality.
- Physical habitat removal / modification-in-stream.
- Other exotic fish.
- Barriers to fish passage.

We chose these on the basis of some of the issues raised in reports we reviewed, or based on our own knowledge of the significance of various stressors on Australia's waterways.

We identified four main criteria on which comparisons could be made. These were:

- ***Scale of impact***, which covers both spatial scale and temporal scale;
- ***Impact type***, which covers impact mechanisms such as predation, competition and habitat alteration and impact consequences, such as increased susceptibility to infection, decreased reproductive output and altered genetics of native fish stocks;
- ***Manifestation of impacts***, which covers altered species composition, the decrease in relative abundance of iconic species and threats to the conservation of endangered species, and

- ***Consequence for management*****,** for which, we considered impact reversibility as the key criterion. For the latter, reversibility for some of the stressors being benchmarked may not be considered pragmatic at all locations where they are an influence. However, we have based reversibility on what is theoretically possible rather than what is pragmatic for the purposes of this exercise.

All comparisons are based on an entirely qualitative (narrative) basis, so it is not possible to formally rank the various stressors in terms of severity of impact. However, the benefit of this approach is that it allows the reader to better understand the nature of the potential impacts of established ornamental fish and where each sits in relation to impacts of other stressors based on information presented for each comparison criterion for each listed stressor.

8.3 Results

Table 8.1, below, provides a summary of the comparisons between the impacts of established ornamental fish and that of other environmental stressors that affect Australia's waterways.

In terms of spatial scale, all the stressors used in this benchmarking exercise occur at discrete locations, though it is probably fair to say that degradation in water quality, altered flow regimes and fishing pressure probably extend their influence over a much larger area of Australia compared with the collective influence of established ornamental fish on native fish. Certainly, these stressors are manifested in all states of Australia, whereas, the influence of established ornamental fish does not currently extend to Tasmania.

In terms of temporal scale, most of the stressors compared in this benchmarking exercise have the potential for ongoing influences on native fish, though it is also difficult to make generalisations about this as, in some particular cases, their influence may be more acute. There may also be cases where their influence is either enhanced or reduced for certain periods. In terms of the potential influence of ornamental fish on native fish, one would expect there to be at least some ongoing influence as long as those species remain present and their effects on the native fish community they interact with is not benign. However, disturbance events, such as flooding, may reduce the populations of some established ornamental fish species with limited tolerance to high flow conditions, thereby reducing their impacts on those native fish communities for a period of time (e.g. *Gambusia* in western Australian streams and in rivers of the Lake Eyre Basin). Control and eradication activities targeting established ornamental fish may also reduce their influence on native fish for short periods (though some methods have the potential to impact native fish at the same time). The corollary of this is the situation where the influence of established ornamental fish on native fish may actually increase during spawning times (if the species in question exhibits aggressive territorial behaviour), or where a species undergoes a rapid increase in population size at a given location (thereby increasing the likelihood of interactions with native fish).

In terms of the other stressors used in this benchmarking exercise, altered flow regimes and degraded water quality are the most likely to have the potential for affecting native fish over discrete time intervals. Degradation in water quality, particularly elevated nutrients, turbidity, and decreased oxygenation, can occur as pulse events associated with heavy rainfall, though they can also occur as chronic disturbances. In terms of flow alteration, water may be stored and released from reservoirs at fixed time intervals, sometimes as a way of mimicking natural environmental flows, though the pressure from growing populations and expansion of agriculture in some areas and also results in a more chronic flow reduction.

In terms of mechanisms of impact, only the stressors involving the introduction of alien species (including translocated species) have the potential to directly impact upon native fish species via the full range of impact mechanisms covered in this benchmarking exercise (albeit, that the introduction of truly exotic fish species has a very low likelihood of having direct genetic impacts on Australian native fish). Of the remaining stressors, habitat removal/destruction and degraded water quality have the potential to impact native fish via a range of mechanisms. The impacts of altered flow regimes on native fish are likely to be indirect effects in most cases.

In terms of the manifestation of the impacts of the various stressors being compared, all have the potential to alter species composition, though the mechanisms for this may vary between stressors. The potential to cause a decline of iconic or threatened native fish species is potentially associated with virtually all the stressors covered as part of this benchmarking exercise, though there is a general need for more information to be gathered before such potential impacts can be confirmed for many of these stressors. In some cases, there is no obvious potential impact mechanism either.

In terms of the key criterion, reversibility, there is a much greater potential for reversibility for environmental stressors that are not linked to the introduction of alien species, even though there will always be instances where there are limited options for this, or amelioration of these impacts is not totally practical. Innovative technologies and improved ecological understanding of the mechanism of impact have certainly made reversing the effects of stressors such as altered flow regimes, degraded water quality, loss or removal of aquatic habitat and barriers to fish passage much more feasible. Reversibility of the impacts of alien species, including established ornamental fish, is thought to be exceedingly difficult, except for some species at very local scales with the aid of control and eradication programmes. Even then, reversibility is not guaranteed, or may only be for a certain time period (e.g., carp in Tasmanian lake systems where rotenoning was carried out in the early 1970's- (Bomford & Tilzey 1996). Eradication of alien fish species is considered virtually impossible by many workers and, as with mitigation measures for other stressors, may not always be practical. Emerging control and eradication measures may eventually improve prospects of reversibility of impacts on native fish associated with established ornamental fish, or at least greatly reduce those impacts, so research effort should be invested in this area in the future.

Table 8.1: Summary of the benchmarking exercise comparing the impacts of established ornamental fish species with other well-known environmental stressors that impact on Australia's waterways.

		Aquarium fish	Other exotic fish	Flow regime	Water quality	Fish habitat	Barrier to fish
Scale of Impact	Spatial scale	Expanding as these species increase their range and as new introduced aliens become established.	Discrete locations, but an impact that occurs to a degree in many parts of Australia. For salmonids and carp, mainly in the south-eastern region. For Gambusia, mainly the northern region.	Don't know. Might stay the same or reduce due to current awareness of water use and environmental flows.	Likely to expand as population grows and the process of urbanisation and agricultural expansion continues.	Discrete locations, but an impact that occurs to a degree in some form in many parts of Australia.	Discrete locations, but an impact that occurs to a degree in many parts of Australia.
	Temporal scale	Ongoing, can be disrupted by environmental changes, such as flooding, or enhanced during spawning or sudden population explosions.	Ongoing, but degree disrupted by environmental changes, such as flood.	Depends on species and type of flow alteration. Where flow release is regulated, impacts might be continuous or discrete depending on the species and their spawning and feeding habits.	Ranges from pulse events through to press (persistent).	Until remediation occurs, impacts are ongoing.	Ongoing unless floods occur that enable barriers to be bypassed.
Type of Impact	Predation	Perceived for some species.	Perceived for salmonids in particular, but also for Gambusia on eggs of native species.	No direct effects, but indirect effects are possible.	Perceived – degraded turbidity could affect predator-prey relationships among species that rely heavily on visual senses to find food or escape predators.	Yes – removes feeding and shelter habitats.	No direct effects, but exclusion of some species may mean decreased predation for other species upstream of barrier.
	Competition	Perceived for which species.	Perceived – particularly for species that overlap in diets or the region of the water column they occupy. Gambusia territoriality is an example of the latter.	No direct effects, but indirect effects are possible.	Yes – suspected that alien species have a competitive advantage over native species under degraded water quality conditions.	Yes – reduced habitat would mean more competition for space.	No direct effects, but exclusion of some species may mean decreased competition pressure for other species upstream of barrier.

		Aquarium fish	Other exotic fish	Flow regime	Water quality	Fish habitat	Barrier to fish
Type of Impact	Fish health	Perceived for which species.	Yes – e.g., disease associated with gold fish.	No direct effects, but indirect effects are possible. For instance, if fish are in poorer condition as a result of flow alterations, they might be more at risk of infection.	Yes – for example, acid sulphate runoff is thought to be linked to the increase in the prevalence of red spot disease among native fish.	No direct effects, but indirect effects are possible. For instance, if fish are in poorer condition as a result of flow alterations, they might be more at risk of infection.	No direct effects likely.
	Reduced reproduction	Perceived where density-dependent impacts affect rare species.	Yes – Gambusia consumption of eggs of other small natives.	Potentially – could affect fish that require certain flow volumes or higher flows at specific times to trigger spawning or migration.	No direct effects, though thermal pollution might effect spawning activities.	Yes – removal of snags means loss of surface to lay eggs for some species. Likewise, the smothering of coarse sediment habitats by fine sediment means loss of spawning habitat for some species.	Yes – restricted access to spawning areas by some species.
	Genetic effects	Not likely, but density-dependent effects on rare species genetics may occur. Habitat fragmentation and reduced gene flow.	Not likely, but density dependent effects on rare species genetics may occur.	No direct effects, but possible density dependent effects on rare species genetics may occur.	No direct effects, but possible density dependent effects on rare species genetics may occur.	No direct effects, but possible density dependent effects on rare species genetics may occur.	Potentially – reduced genetic flow between upstream and downstream populations.

		Aquarium fish	Other exotic fish	Flow regime	Water quality	Fish habitat	Barrier to fish
Effect of Impact	Change in species mix	Yes – as a new species is being introduced and there may also be changes in species composition of native fish communities as a result of impacts associated with introductions.	Yes – as a new species is being introduced.	Yes – give examples – though a better understanding of the impact mechanism and impact consequences associated with this stressor is needed to further support this assumption.	Perceived – based on information presented for impact type. However, it is sometimes difficult to disentangle the impacts of degraded water quality with those associated with alien fish, or other stressors.	Yes, via loss or reduction of species that rely on those habitats.	Yes – upstream and downstream species composition will be different and downstream community would change as a result of a decline in the populations of affected species over time.
	Reduced iconic species	Perceived for some species.	Perceived – e.g., Murray Cod stocks could be reduced by disease.	Yes – give examples – though a better understanding of the impact mechanisms and impact consequences associated with this stressor is needed to further support this assumption.	Possibly – need more information on direct linkages.	Yes – desnagging and its effects on the Eastern Cod (Andy Moore pers. comm.)	Yes – several species.
	Reduction in populations or range of threatened, endangered and vulnerable species	Perceived, but more evidence is required.	Perceived threats to numerous species of galaxiids and several species of pygmy perch.	Yes – give examples – though a better understanding of the impact mechanisms and impact consequences associated with this stressor is needed to further support this assumption.	Possibly – need more information on direct linkages.	Yes – desnagging and its effects on the Eastern Cod (Andy Moore pers. comm.)	More information needed – some barriers to Australian Grayling in Victoria have been modified to increase fish passage specifically for this species (Jacques Boubee, NIWA pers. comm.)

		Aquarium fish	Other exotic fish	Flow regime	Water quality	Fish habitat	Barrier to fish
Management	Reversibility	Only on a very local scale, but almost impossible over large scales. Even for small scales, reversibility isn't guaranteed.	Only on a very local scale, but almost impossible over large scales. Even for small scales, reversibility isn't guaranteed.	Reversibility is possible through removal of dams, or altering the timing and volume of environmental flow release based on the requirements of native flora and fauna strongly affected by flow regime.	A degree of reversibility is possible for most activities that lead to degraded water quality. Technologies for ameliorating water quality will continue to emerge also. Reversibility might be limited by population growth and pre-emptive use of land.	Some degree of reversibility is afforded through actions such as replacing riparian vegetation and snags and reducing sedimentation.	Opportunities for reversibility reasonably good, through removal of barriers altogether or replacing or modifying them so that fish passage is improved.

9. Overview of control and eradication methods for pest fish

9.1 Introduction

Eradication of pest fish is desirable but is rarely feasible and it may not be an essential part of managing a pest fish species. This is especially so where impacts may be partially related to other stressors and removal could result in little measurable improvement. If eradication of a particular species will be expensive and cannot be shown *a priori* to result in any ecological or social benefit, then managers may opt to do nothing. Similarly, if the exotic fish species is known to have negligible impacts then there is little point in implementing control programs, particularly if these are costly and need to be repeated, or if they are not considered by the general public to be socially or economically acceptable. A danger with this approach is that impacts may arise later if the environment changes, or if the species is later spread to other environments where conditions are different and where impacts do occur (Simberloff 2003; McDowall 2004). If this possibility is accepted, then resource managers cannot accept the 'do nothing' approach and, as a minimum, need to ensure that any further spread does not occur.

Eradication is generally taken to mean the complete removal of exotic species from a defined area but this needs to be further qualified by a given time frame. For example, the successful removal of carp from lakes in Tasmania occurred over a 20 year period and was considered a successful eradication campaign, even though the species was re-introduced later. Hence Bomford and Tilzey (1996) considered that when eradication is the management goal, it should be time-limited. This definition implies that resource managers need to set achievable time-bound targets for the management of pest fish species in order to provide a clear indication of the intent and costs of management.

Where eradication is not an option, the main objective for resource managers is to reduce the impact of pest fish species to an acceptable level. However, defining an acceptable level of impact requires a good understanding of the impacts as well as identification of the relationship between these and pest fish densities. This step is often overlooked in pest control programmes because of the need to act quickly combined with the high cost and long time frame needed for research to quantify such relationships. However, such research can be important where other variables are contributing to the impacts created by pest fish and so confound their role. Where this occurs, the effects of pest fish control alone may be limited. Such research is also needed to establish baselines for both fish density and key environmental variables so that the effectiveness of the control programme can be assessed.

Because of the cost and time involved in carrying out the preliminary research needed to properly assess the effectiveness of control programmes, an adaptive management approach is often adopted. On-going control measures such as netting are carried out to reduce pest fish densities and key environmental variables are measured concurrently to determine the environmental response. Such management experiments can be extremely useful if carried out under scientific supervision so that they can also provide a *de facto* manipulation experiment. Manipulation experiments are a key tool for identifying the true impact(s) of pest fish (see Chapter 3), but they require knowledge of fish densities. A major limitation of the adaptive management approach to pest fish control is that while the rate of fish removal can be measured, fish density is generally not, so the relationship between fish density and impact level cannot be determined. This leaves managers in the unenviable position of not knowing what level of control needs to be maintained. Methods for assessing fish density therefore need to be grafted onto such control programmes to enhance their value and to help indicate what level of control is acceptable.

When considering the feasibility of eradication or control programs, the costs imposed by the impacts of the introduced fish on the environment and the community need to be compared with the costs involved in the pest fish management program, as the latter can be prohibitively high. For example, Jackson et al. (2004) noted that one of the practical limitations of effective impact management is the generally high labour and economic cost of management methods. They suggested that a strategy to eradicate Johnson's Lagoon trout would involve *"78 person-days, 51 person-nights, 4800 km travel, with follow-up monitoring required to ascertain the success of the operation and to detect new introductions."* In comparison, the economic cost of efforts to control and eradicate carp in Tasmania over a 20 year period will have been orders of magnitude higher than this. This cost-benefit issue is often a matter of scale and hence of the size of the environment(s) being considered for treatment. Eradication in a small closed system may be feasible, cost effective and require little time, but in a larger closed system it may be uneconomic even if feasible over the long term. Eradication is rarely considered in open systems because it is generally not possible, let alone economic. A further issue with cost-benefit comparisons is that environmental costs and benefits are not easily measured and expressed in dollar terms and so cannot be readily compared with the economic costs of fish control. Judgement is required to make this comparison and this requires a clear appraisal of the ecological impacts, plus the consequences doing nothing as this could allow further damage to occur, along with a good estimate of the costs of control.

The difficulty in comparing ecological impacts with the costs of control means that social factors can play a large role in the decision to undertake eradication or control. For example, acceptance of the type of control method by the public may be an important issue in large public water-bodies, especially those that are intensively used. The public may have an aversion to the use of some chemical methods and to the collateral damage to other wildlife. There may also be an objection to the long time-frames for control, especially if control methods will compromise other uses of the waterbody. These sorts of issues reflect the different priorities of water users and they need to be resolved alongside cost/benefit considerations through public consultation.

Animal health and welfare issues also need to be considered. The RSPCA believes that the general principles for the control of introduced vertebrates as stated in their policy (see below) should apply to the control of exotic fish. These principles were developed by the Humane Vertebrate Pest Control Working Group in 2004.

> 'RSPCA Australia recognises that wild populations of introduced animals can adversely affect natural ecosystems, endanger native plant and animal species, jeopardise agricultural production and can harbour pests and diseases. RSPCA Australia acknowledges that in certain circumstances it is necessary to reduce or eradicate populations of some introduced animals. The killing of introduced animals should only be sanctioned where no successful, humane, non-lethal alternative method of control is available. Any measures taken to reduce or eradicate specific populations of introduced animals must recognise that these animals require the same level of consideration for their welfare as that given to domestic and native animals. Control programs must be proven to be necessary and potentially successful at reducing the adverse impact of the target animals. Such control programs must be conducted humanely, and be under the direct supervision of the appropriate government authorities. They should be target-specific, not cause suffering to non-target animals, and should be effectively monitored and audited with resulting data made available for public information. RSPCA Australia opposes the commercial removal and use of introduced animals unless such use is carried out in a humane manner and only as part of a fully regulated government supervised management program. Commercial operations should not be permitted to sustain population levels of these animals to the detriment of the environment and the animals involved.'

Another important social factor will be the likelihood of re-introduction and the feasibility of measures to prevent this. Where successful eradication or control will be thwarted by clandestine re-introduction(s) of exotic fish, then it is pointless to carry out such management until the risk of re-introduction can be reduced. Education based on solid evidence of harm is required to target the proponents of re-introduction and to reduce this risk before eradication or control can be implemented. In some cases, this may take a generation to occur as some proponents may be unable to change their views and a reduction in the risk of re-introduction will then depend on education of the next generation.

It has already been noted (Chapter 3) that control strategies for ornamental fish species now present in the wild in Australia may be either site- or species-led, depending on the extent of their distribution and the locations of wild populations. The choice of control strategy also depends on the method of control that can be applied to each species. A range of control and eradication methods have been used to mitigate the impacts of exotic fish species in both Australia and abroad, though few of the 23 listed established ornamental fish covered in this report have been the subject of these. The following chapter therefore reviews these methods and their application and notes the lessons learnt that can be applied to ornamental fish.

The various control and eradication methods fall into five broad categories; (a) physical removal methods, (b) chemical methods, (c) biological controls, (d) habitat manipulations and (e) genetic and biochemical methods. Often, more than one type of method needs to be applied simultaneously. This is particularly true for chemical and physical removal methods. However, this chapter is not intended as a prescription of what methods to use for which species in what places. Experience has indicated that the type or combination of methods can vary greatly depending on site and species-specific factors. Thus, this chapter reviews the potential choices of method that can be potentially used to control and in some cases eradicate exotic fish. Some of the methods are still classed as experimental in that they have not yet been applied, however, the high level of public awareness of their potential means that some comment on their potential use is required.

9.2 Physical removal methods

Netting, trapping, line fishing: These methods are proven techniques for removing fish, but are typically only considered as control options because their application needs to be repeated. These methods often require intensive effort to be effective and their application is often limited by factors such as access, water depth, water velocity, aquatic plant cover, logjams and the development of avoidance behaviour by the targeted species. They are often invoked where other more effective methods of control are not practical or not supported (Mick Holloway, NSW, pers. comm.).

One of the main drawbacks associated with these methods includes the high overall cost of repeat treatments, particularly in circumstances where it is difficult to restrict the re-introduction of the target species into the treated area. There may also be social acceptability issues related to both the use of humane ways of capturing and disposing of the fish and to the impacts of netting on other fauna.

If the task of removal by netting, trapping or fishing is given to commercial harvesters rather than being undertaken by government or state agencies, there is the potential that boom-bust cycles will eventually discourage industry participation over the long term and, therefore, the potential for long-term control will be compromised. There is also the potential for vested interests within the commercial harvesting business to encourage the further spread of the exotic species as a way of maintaining a continued supply of fish and hence of income. If commercial harvesting is to occur, stringent management protocols would need to be put in place to ensure that harvesting can be economically sustained in the long term, and that further spread of established ornamental species is prevented. It will also be necessary to determine whether the economically sustainable level of fish harvest results in a quantifiable reduction in impacts.

Gill netting can be used to reduce the density of some of the larger pest fish and to thereby reduce their density and impact, but it is rarely sustained as a control method because of the high labour cost involved. Gill netting is selective and tends to work much better on larger species than on smaller species. Another potential risk associated with gill netting is that there will be collateral damage to other species. In addition, there may also be bio-security concerns if nets are not cleaned properly and are used in different water bodies, resulting in the potential spread of pest species. Another unexpected consequence of netting is that selective capture of large piscivorous fish can sometimes promote population growth of the targeted species by limiting predation on juveniles.

Beach seining and purse seining are used to target aggregations of fish in shallow surface waters and may be effective on small fish in the shallows provided obstructions such as weed, rocks and logjams are not present. Seine netting was the main method used to reduce carp in Gippsland lakes (Bell 2003)

Trapping is generally used to capture fish undertaking migrations to or from spawning habitats. Traps have been recently devised to catch migrant common carp in streams by forcing them to jump over an artificial barrier into a holding pen (Stuart et al. 2003). Netting was successful in reducing carp abundance in Lakes Cresent and Sorrel in Tasmania, but eradication is proving more difficult and whereas it may be possible in Lake Cresent, it may not succeed in the much larger Lake Sorrel (ASFB 2005). Fencing is now being used in conjunction with traps to prevent carp spawning and to enhance carp capture in traps in these lakes (Diggles et al. 2004). Radio tracking studies have revealed that most carp migrate through a narrow isthmus on one side of Lake Sorrell to reach spawning grounds on the other side and this presents an ideal opportunity for trapping (ASFB 2005).

Line-fishing is a proven technique for the removal of the larger fish and in Australia, 'Carp Watch' members are the only known collective that targets alien fish species using line-fishing as part of a conscious control effort[7]. Their effort is restricted mainly to the Murray-Darling system at present. Line fishing works only for larger fish and hence is not for small-bodied species. Effectiveness is also governed by the extent to which the exotic species targeted is likely to take baits or lures. Line-fishing is not thought to be an effective control or eradication option in its own right and is more likely to be undertaken by members of the public than government agencies. If anglers are going to support line-fishing as an exotic fish removal technique in Australia, it will be only for those species known for their size and/or 'fighting' quality. Whereas *Tilapia,* Oriental weatherloach and Oscars may exhibit such behaviour, it is unlikely that many of the other established ornamental fish will have such traits. Therefore, line-fishing is a technique that probably has only a limited application for removal of established ornamental fish in Australia. With the public undertaking line fishing of a designated 'pest fish', there is always the risk that anglers

[7] Carp Watch is the only group dedicated to the recording and removal of carp from Australian waterways.

may not always dispose of fish in a humane way. However, a greater risk is that anglers targeting exotic species for recreation (with control as a secondary motive), may wish to spread them further to provide more recreational opportunities.

Bow-fishing is used by bow hunters in New Zealand to target koi carp a variant of common carp in the Waikato River. Annual competitions can result in the removal of many large fish, but this effort is unlikely to have any significant impact on the overall population.

Although it is unlikely that recreational fishing will ever reach levels where it could be considered as a control option in its own right in Australia, it could be part of the arsenal of control measures for some of the listed established ornamental fish species. Tilapiine species and Oriental weatherloach are most likely to be targeted. For example, removal and disposal of tilapia is part of the annual ‘Barra bash’ in Lake Timaroo, and several tilapia removal fishing events have been held in the Mulgrave River in northern Queensland (pers. comm., Brett Herbert). Oscars are also known to be prized game fish, but this species has a very narrow distribution range in Australia, being restricted to a cooling pond for a thermal power plant in Victoria. If recreational fishing for pest species is to be an activity supported by resource management agencies, then education programs may need to be put in place to educate anglers about humane ways of capturing and disposing of captured fish as well as to underline the dangers of spreading these species.

Electric fishing and explosives: In general, electrofishing is the most cost efficient physical method of fish removal in shallow waters and is capable of removing a wide range of fish sizes. Electrofishing has been used in the management of carp in waterways in NSW (Mick Holloway, NSW Fisheries, pers. comm.) and control of tilapia by the Queensland Department of Primary Industry (pers. com. Brett Herbert). Electrofishing from boats is generally constrained to waters less than 3 m deep and is a potentially useful method for reducing pest fish, but not for eradicating them. Repeat electric fishing in small streams has been used to eradicate small fish living above natural or man-made barriers (e.g., above a water fall or a weir) (e.g., Lintermans 2000) but eradication is unlikely to be possible in larger systems, or in streams where water depths exceed about 1 m and where instream cover provides refugia from electrofishing.

Following a reduction in water level, explosives were used three times by the New South Wales DPI to eradicate a population of Jack Dempsy in a pool of a disused quarry in Angourie (Mick Holloway, NSW Fisheries, pers. comm.; ASFB 2006). Explosives can be useful in small water-bodies where the ‘effective’ blast field can encompass the entire water mass. However, explosives have not proved effective in large, deep water bodies (Pullan 1982). This is because the ‘effective’ blast field is spatially limited and in large water bodies it may be impracticable to set enough charges to provide complete coverage. Even the extensive cover provided by the use of detonation cord and power gel explosives in the Angouri quarry may not have eliminated the Jack Dempsey cichlid because this species has been recently found there again.

Water removal: Pumping water out of ponds, small lakes and water holes allows the easier removal of fish by physical and or chemical means and, where habitats can be pumped dry, eradication may then be achieved without additional methods. In 2001, this method was utilised to eradicate *Gambusia* from a pond in Todd Mall in Alice Springs. The size of this waterway is unknown, however, the method was considered completely successful for eradicating this species in this water body (Australian Broadcasting Corporation, 2001). Gambusia were also eradicated from the Ilparpa Swamp and from three ponds on residential properties in Alice Springs (ASFB 2003a). The swamp was drained by pumping and evaporation then resulted in desiccation and the removal of all fish.

In New South Wales, pumping down of a waterway was used in conjunction with explosives to eradicate a population of Jack Dempsey in a pool that had formed within a disused quarry in Angourie. The eradication of Jack Dempsey required three attempts before it was successful. It was estimated that the Jack Dempsey eradication involved three person days as well as the cost of contracting an explosives expert to undertake the eradication. It also involved pre- and post-survey work (Mick Holloway, NSW Fisheries, pers. comm.)

Drawdown of water generally involves the removal of remaining fish from the residual pools by physical or chemical means, and this can mean that non-target species can be salvaged and kept alive for later restocking. It can be an expensive method in large water-bodies but can work well for a wide range of fish species and size classes, especially in conjunction with other methods. It is not feasible in water bodies where inflows cannot be diverted or dammed.

A major limitation of this method is the ability to safely dispose of the pumped water. If water intakes cannot be screened or filtered to remove larval and small juvenile fish, then the water needs to be sprayed overland to ensure that larvae and juveniles are not carried into downstream waterways. This can be a major issue in large water bodies where large amounts of water need to be disposed of over a short period of time (e.g., several days) and where a constant overland flow of water to some natural waterway consequently develops.

Drainage of water will result in the destruction of aquatic macrophyte beds and changes to the bottom substrate, both of which could both have cascading ecological effects on native aquatic fauna and the habitats and ecological processes that maintain them. However, in small static water-bodies this may be an acceptable ecological price to pay for the eradication of the pest fish species.

9.3 Chemical toxicants

Rotenone: The use of rotenone for the control on non-native fish in Australia has been well reviewed by Rayner and Creese (2006). Rotenone is the principal chemical used to control and eradicate exotic fish species in both Australia and abroad. It is a liquid toxicant and is mixed into the water where the target species is present to produce the minimum concentration needed to kill the species. Different concentrations are required for different species and this chemical can be applied in various forms.

Rotenone is the most widely used and popular form of pest fish control and has been routinely used in a number of countries for this purpose for over a century. Records of rotenone application in Australia include the rotenoning of 20 dams in Tasmania in the 1970's, and 1300 dams in Gippsland, Victoria in the early 1960's to control carp. Both programmes were considered successes, though carp were re-introduced to the Tasmanian dams some 20 years later and carp were recorded some 3 years later in the Yallourn storage dam in the La Trobe river system.

Rotenone was also applied unsuccessfully to ponds in Townsville to rid them of mosambique tilapia (Arthington et al. 1984) and to two ponds in residential properties in the Northern Territory to remove populations of *Gambusia* (ASFB 2003a). In a recent operation in NSW, rotenone was utilised to partially eradicate a population of one-spot livebearers from a series of ponds located on the Long Reef Golf Course (Rayner & Creese 2006). In their review of rotenone use in Australia, Rayner & Creese (2006) reported the successful use of this piscicide to eradicate gambusia in twelve pools near Kurnell in New South Wales and in waters near Alice Springs, jewel cichlids from a drainage channel of the Royal Darwin Turf Club, a population of over a million Mosambique tilapia from a pool in Port Douglas, tilapia from 2 ha pond near Ipswich in Queensland, perch from Brushy Lagoon in Tasmania, and trout from small streams ranging from 2.4-20 km long in the Australian Capital Territory and Victoria. Rotenone has also been used to eradicate white cloud mountain minnows from an isolated waterhole in a small creek in Brisbane (ASFB 2003c).

Rotenone application is a highly effective method for the eradication of pest fish in enclosed systems but local conditions can have large bearing on its success rate (Rayner & Cresse 2006). The application of this chemical needs to account for the maximum depth of the water body, low water temperatures, high turbidity and exposure to sunlight[8]. Rotenoning is more viable in easily mixed[9], shallow water-

[8] At water temperatures less than 12°C, rotenone use is less effective, while at higher sunlight levels it will remain toxic for weeks Sanger, A. C. and J. D. Koehn (1997). Use of chemicals in carp control. Controlling carp: exploring the options for Australia. Proceedings of a workshop 22-24 October 1996. J. Roberts and R. Tilzey. Canberra, CSIRO and the Murray Darling Basin Commission**:** 37-55.

[9] In some cases, fluorescent dye has been used to determine whether effective mixing has occurred (e.g., the Victorian stream application case studies cited in Ibid.. For those studies, riffle zones were used as places for applying the neutralising agent to ensure it mixed with the rotenone in the water column. Boat motors have sometimes been used to help mix the rotenone

bodies where aquatic cover (e.g., macrophytes, wood jams) is limited. When applied in open systems, it is limited to small streams where water flow can be managed to maintain 'effective' concentrations for the time needed to effect a kill (several hours but usually a day in practice). Small enclosed sections may need to be created and treated sequentially while proceeding downstream.

The application of rotenone can result in collateral damage[10] to native species (e.g. other fish and amphibian including turtles) unless salvage and resuscitation operations are carried out concurrently. Fish resuscitation is possible by placing affected fish in clean water. The rotenone can also be neutralised by the addition of potassium permanganate to the water. If populations of the target species are larger than expected, or if there is a high degree of collateral damage, there is the potential for users to become overwhelmed by the large quantities of fish produced. Robust plans for dealing with the removal of a potentially large numbers of fish are required when using this technique (Sanger and Koehn 1997).

Perception issues relating to concerns over use of chemicals in waterways may prevent attempts to use this technique in some instances. Some liquid forms of rotenone have synergists to allow the mixing of rotenone with water and the ecological effects of these may be a concern[11]. At present, there are no supported cases of human health risk associated with using the types of quantities of rotenone required to control alien fish populations at small to medium scales[12].

Rotenone is approved for use in most States but as of 1996, it was not approved in all (Sanger and Koehn 1997). In 1996, only the liquid form was available for use in Australia (Sanger and Koehn 1997). Rotenone use has been recently banned in Victoria on somewhat 'dubious' grounds (ASFB 2005). Legislation in New Zealand now prevents the use of the liquid form as it contains a synergist, whose impacts are yet to be determined. The powder form (derris dust) is now used in New Zealand to avoid introducing chemical synergists into waterways.

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into water columns of shallow closed systems McDowall, R. M. (2006). The truth about rotenone. Fish and Game New Zealand. 51: 61-63.

[10] Though this can be reduced if the native fish are rescued and put into fresh water at the time of application Ibid., or if a neutralizing agent is applied where rotenoning is carried out in stream sections (Sanger, A. C. and J. D. Koehn (19967. Use of chemicals in carp control. Controlling carp: exploring the options for Australia. Proceedings of a workshop 22-24 October 1996. J. Roberts and R. Tilzey. Canberra, CSIRO and the Murray Darling Basin Commission: 37-55.

[11] Though many are similar to those used in household solvent products (McDowall, R. M. (2006). The truth about rotenone. Fish and Game New Zealand. 51: 61-63.

[12] Rotenone breaks down quickly under normal conditions, so its effects aren't likely to be persistent. Another strategy is to apply rotenone (or other chemicals) when water levels are low, to minimize the spread of these chemicals or the need for neutralisation agents to be applied.

It is rare for large quantities of rotenone to be used at one time, though this has been done in other countries, such as the USA[13]. Rotenone has generally been applied over small areas, though there have been notable exceptions to this in other countries[14].

One potential limiting factor in the success of rotenone application for pest fish control is that the organisations that approve the use of rotenone and those that apply it are often different. Where an urgent need for control occurs, this difference can result in unacceptable delays. This situation occurred when a population of carp was first found in the Glenelg River (ASFB 2004b). Sanger and Koehn (1997) have therefore advocated that robust risk assessments and communication plans are prepared before rotenone is applied, with contingencies for emergency eradication situations[15]. Potassium permanganate is sometimes used to neutralise rotenone and reduce the time needed for it to degrade naturally. This reduces the time before restocking of desirable species can occur.

Baits containing rotenone or antimycin have been recently developed to allow the targeting of pest species (e.g., Mallison et al. 1995; Kroon et al. 2005), thereby reducing the risk of collateral damage. This method is still experimental and allows for control, but not eradication. In time, further refinement can be expected to allow this method to become more effective and better targeted such that it can be used as a viable control method.

Antimycin: Antimycin is a stronger toxicant than rotenone but has not been used extensively as yet. Its application is constrained by much the same considerations as those applying to rotenone, but fish recovery is usually not possible. Sanger and Koehn (1997) reported that antimycin was not available in commercial quantities for use in Australia in 1996. They also stated that the local production of this chemical in Australia may face problems in terms of negotiating with the patent holder for the right to do so.

Agricultural pesticides: The use of agricultural pesticides such as acrolein and endosulfan is regarded as experimental as they have not been used extensively in Australia as yet. Furthermore, neither acrolein, nor endosulfan were registered as piscicides in Australia as of 1996 (Sanger and Koehn 1997). The dose rates also require further clarification (Sanger and Koehn 1997). As with other chemical dosing techniques, these chemicals are more likely to be viable in well-mixed, shallow water bodies. However, these chemicals are far more persistent in the environment than

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[13] 20 tonnes was used in a single reservoir in Utah (cited in McDowall, R. M. (2006). The truth about rotenone. Fish and Game New Zealand. 51: 61-63..

[14] A 400km stretch of river in Russia and a 700 km section of river in California were treated with rotenone (cited in McDowall 2005).

[15] Sanger, A. C. and J. D. Koehn (1997). Use of chemicals in carp control. Controlling carp: exploring the options for Australia. Proceedings of a workshop 22-24 October 1996. J. Roberts and R. Tilzey. Canberra, CSIRO and the Murray Darking Basin Commission: 37-55.

rotenone (Sanger and Koehn 1997), so there is a far greater risk of long-term, adverse environmental impacts ranging from mortality through to bioaccumulation.

Lime: Liming with calcium hydroxide produces a high pH and is an established chemical control in small, closed, easily-mixed, water-bodies, particularly ponds where access by wildlife and members of the public can be prevented for the duration of treatment. The main advantage over rotenone is cost and availability. However, liming raises the pH to over 10 and the resultant caustic water poses a threat to wildlife as well as a health & safety risk to humans. As with most other chemical dosing techniques, collateral damage to native species is high.

Lime was added to some waterways affected by carp in Victoria in the early 1960's. It was considered to be effective at the time even though only half of the reported numbers of stocked carp were recovered. Divisional officers reported satisfactory results (Barnham 1998). Lime was also used to control populations of *Gambusia* in NSW (NPWS 2003) and in Tasmania (ASFB 2005). The Inland Fisheries Service applied lime to a dam near the town of Snug to eradicate *Gambusia* but this was unsuccessful even though the pH was raised to over 11. In larger environments, it is more difficult to mix chemicals throughout the entire water body and there are more opportunities for fish to find refugia.

Chlorination: Chlorine dosing with solutions of calcium/sodium hypochlorite is, like lime dosing, an established viable chemical control in small closed water-bodies, and it is used in the same places where lime dosing can be applied. It is similar to lime in terms of the high likelihood of collateral damage to native fish and the potential to represent a human health hazard. It was used to control populations of *Gambusia* in NSW (NPWS 2003). In the Northern Territory, chlorine was used to eradicate a population of platys, which had become established in a storm water drain in Alice Springs. This operation was undertaken during the dry season so that the drain was a closed system and did not flow into other waterways. The cost of the method involved 2 person days and the purchase of a drum of chlorine. No other species were apparent and there was therefore no collateral impact on other species. Chlorine was utilised extensively in the eradication of the black striped mussel in coastal waters of the Northern Territory. This involved over 300 personnel and it included the tracking and treatment of shipping vessels that had left infected sites, plus the treatment of three sites and almost three hundred vessels in the Darwin area, and the initiation of a public awareness program. The total response effort was costed at over $2 million (Macauly 2000). The scale and costs of applications of chlorine for pest fish control in freshwater systems is likely to be far less than that for the black striped mussel in Darwin Harbour but application will have a greater degree of collateral damage to both other organisms and the environment than rotenone. Its major advantage is its cost and availability.

9.4 Biological controls

Introduced predators: The introduction of predators to reduce pest fish is considered an experimental rather than a proven method at present because it is yet to be widely demonstrated. It is also a control rather than an eradication method because predators are highly unlikely to drive a prey species to extinction, except in very small and simple environments lacking refugia. There have been various calls to introduce native fish predators to control exotic fish (e.g., Murray Cod and shortfin eels to control common carp in the Glenelg River – ASFB 2004a, and for the restoration of native piscivores to the upper reaches of rivers where aquarium fish now occur in degraded habitats –ASFB 2003b), but there are few instances where this has occurred. Australian bass were introduced to a waterway in New South Wales to control a wild population of Jack Dempsey. The costs involved in the sourcing of the introduced predator were not high as the bass were being bred in the agency's hatchery. Bass were also prevalent in the geographical location of the interaction (Mick Holloway, NSW Fisheries, pers. comm.) so escapees were not an issue.

To be effective, piscivores known to consume the target species, or at least to be capable of feeding on that species, need to be identified. In addition, the effectiveness of piscivorous fish will be governed by the degree to which the target pest fish species exhibits anti-predator behaviour[16], how fast it can reproduce (i.e., how resilient its populations are likely to be to mortality through predation), the abundance of alternative prey species, and the prevalence of refugia for the prey species. Species of ornamental fish that exhibit anti-predator behaviour, such as certain cichlids and poeciliids, or those species with a very high resilience due to their high reproductive outputs, are less likely to be vulnerable to control by the introduction of predators.

Australia does not have many large, native, piscivorous predators (Koehn 2004) that could potentially be bred and made readily available for control programs, so other exotic species may have to be identified, bred, made infertile, and then used for this purpose. Choosing a predator species that is likely to be both effective for the purpose of its introduction, low risk in terms of potential ecological impacts, and easily removed or reduced once control has been achieved could prove problematic. There is always the potential for unforeseen impacts to arise with introduced predators, including greater impacts on native species. To avoid any long-term, unacceptable damage to native fauna, the introduction of a fish predator may require a species that will not breed in the target environment, or fish stocks that have been sterilised. This means that periodic stocking will be required to maintain control over the pest population. Alternatively, stocking can be halted to allow re-establishment of the status quo.

[16] There are several species that exhibit anti-predator behaviour including schooling, hiding and responding quickly to chemical cues or distress from con-specifics (e.g., midas, cichlids, and guppies). These species are less likely to be suitable for control using this particular method.

Members of the public and resource management agencies alike are likely to be very wary of predator control because of Australia's experience with the cane toad, *Bufo marinus*, which was introduced into Australia as a predator to control the cane beetle. Due to the potential risks associated with this means of control, it is unlikely to be suitable for application in open systems, so is only likely to be considered as an option for certain established ornamental fish species in closed systems. For example, there is good evidence that a piscivore (bass) controlled *Gambusia* in an Australian lake and dam (A. Moore, pers. comm.) and that rainbow trout controlled *Gambusia* in a New Zealand lake (Rowe 2003). Stringent risk management plans, not unlike those put forward for rotenone use by Sanger and Koehn (1997), should be put in place whenever this method is considered.

Introduction of pathogens: The introduction of fish pathogens (e.g., parasites, bacteria, viruses) as a means of controlling or eradicating pest fish species is another method that is considered experimental rather than proven. Fish pathogens are usually specific to a family or even a genus of fish, so this technique can potentially be targeted at the pest species and not other fish.

In Australia, the introduced epizootic haematopoietic necrosis (EHN) virus kills redfin perch (Langdon & Humphrey 1982) and has caused high mortality in some wild redfin perch populations. The introduction of the spring viraemia of carp virus (*Rhabdovirus carpio*) to Australia for carp control has been discussed since the 1970's (Crane and Eaton 1996), but this control method has not, to our knowledge, been implemented here due to concerns raised below. Carp herpes virus (CHV) is reported to kill four out of every five fish it affects in Europe and Asia (Pearson 2004) so whereas its spread is being actively prevented in the northern hemisphere, it may be a potential control agent in Australia where carp are a pest species.

Fishes that live in harsh conditions and that are stressed are more likely to be susceptible to the impacts of pathogens. Effectiveness of pathogens will also be governed by environmental conditions (such as temperature) and might depend on the availability of intermediate hosts. Some viruses can be biochemically modified to be made more virulent, more or less host-specific, or to withstand a greater range of temperatures (Crane and Eaton 1996).

The effects of introduced pathogens on the host species are likely to decline as its populations become more resistant and/or resilient. Effectiveness will also depend on whether or not established ornamental fish populations are immunologically naïve to the pathogen in question. If they are, then introduced pathogens are likely to be much more effective. It may be difficult to assess whether or not this is the case for different wild populations in Australia before deciding whether this techniques is feasible. One of the main arguments against the potential effectiveness of this method will be that pathogens, even if they are initially effective, may become ineffective as the host population gradually acquires immunity to the pathogen.

A long-term risk with introduced pathogens is their potential to become less host-specific and, through mutation, to acquire the ability to infect other native fish species. There is, in the long term, the very real potential that a new pathogen could change and affect the economic viability of Australia's fisheries and aquaculture industries. If such a pathogen developed, Australia would become registered as an 'infected' country and it would make sales of fish to other countries difficult, particularly live produce, which is a high value resource[17].

Many members of the public are likely to have problems with the introduction of pathogens as these organisms are normally associated with negative impacts on human health. Strong social resistance may be encountered when attempting to develop this technique.

9.5 Habitat modification

As with the other biological control methods, this procedure is considered an experimental approach rather than a proven technique. It is only likely to be viable for species with specific habitat requirements[18]. In this respect, it is likely to be a species and location specific type of control measure and may not necessarily be applied successfully for the management of the full range of established ornamental species covered in this report.

To our knowledge, this method has not been applied yet in Australia, nor overseas, but is considered potentially viable because the populations of some freshwater fish that spawn in shallow waters on lake shores have declined following a reduction in water level (e.g., Gafny et al. 1992). Water level manipulation is currently being tested for carp control in shallow waters of the Barmah-Millewa forest (Gilligan 2005).

This technique is also only likely to be viable where spawning habitats can either be altered or removed easily, or where it is practical to restrict the spawning migrations of established ornamental fish in a way that does not restrict that of native species, or alter natural flow regimes or ecological processes.

The development of this control option will depend on the identification of key habitats and this reinforces the need for more data on the habitat requirements of many of the 23 established ornamental fish before this technique can be considered.

Both Arthington et al. (1983) and Webb (1994) found that a number of aquarium fish species in northern Queensland waters were thriving in waters where degradation of the habitat had occurred through urban development. Development including, removal

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[17] This would probably be the case if the spring viraemia of carp virus were introduced into Australia for carp control (Crane & Eaton, 1996).

[18] There were several species of established ornamental fish that do have certain requirements for spawning, including the need for fish passage during migration and specific substrates (e.g., *Tilapia mariae*). These populations of these species may be able to be controlled to a degree using this control method.

of riparian trees, increased siltation of substrates and increased nutrient inputs, all served to expose streams to increased macrophyte growth and stagnation, which disadvantaged native fish but assisted the survival of exotic fish. As a consequence he advocated habitat restoration to change the balance between exotic and native fish species. Replacement of riparian planting to decrease stream water temperatures and reduce macrophyte growth can be expected to improve conditions for native fish species while reducing them for aquarium species (c.f., Arthington et al. 1990)

Pritchard et al. (2004) have also advocated habitat manipulation to restore the balance between native and exotic fish species. They observed an increase in native species and a decline in gambusia in rivers of the Lake Eyre Basin in wet years and the opposite in dry years. They attributed these changes in fish abundance to habitat changes. In wet years, the restoration of river flows resulted in the removal of disconnected, isolated pools favouring gambusia and increased their exposure to native piscivores.

However, such habitat modification or restructuring could potentially have unforeseen and even cascading ecological impacts on other fish. Some understanding of the potential consequences for native fauna and flora communities of undertaking this control method should therefore be obtained before this approach is considered.

9.6 Immuno-contraceptive control and genetic techniques

As with biological control methodologies, these methods are also considered to be experimental rather than proven techniques. While both techniques have the potential to reduce populations of pest fish species through a reduction in their reproductive output, reductions in fertility can sometimes be compensated for by greater survivorship of juveniles through lower levels of intra-specific competition. Thus, a high level of fertility reduction over time may be required before any major effects on abundance are realised (Hinds and Pech 1996).

Baits have been suggested as a vector for dispersing immuno-contraceptive drugs, but this depends on the prior development of species-specific baits that are more attractive to a wide range of the target species than their natural prey. The recent issues and concerns over the increase in phytoestrogens in some natural waters is likely to raise public concern over the use of this method.

Genetic techniques involving the insertion of genes resulting in single sex progeny are likely to be highly species-specific, so this technique has an extremely negligible risk of collateral damage to native fish. There is a large amount of research currently focussed on the development of a ‘daughterless carp’ gene in Australia. However, attempts to introduce such a gene into *Gambusia* to demonstrate the viability of the method were not successful, so its application to ornamental fish such as poeciliids may not be possible. Should the method prove viable for other species, there is likely to be some opposition to the insertion of genes resulting in single sex progeny, especially given the current opposition to the distribution of genetically engineered organisms into the wild from some sections of the community. Stringent risk management plans, not unlike those put forward for rotenone use by Sanger & Koehn (1997), should be put in place whenever this method is considered.

9.7 Summary of control and eradication options

There is a wide range of potential options for the control and/or eradication of established ornamental fish, but many of these are currently being developed, or are untried, whereas others all have some drawbacks and limitations in terms of which species they can be successfully applied to, the types of water bodies they can practically be deployed in and their relative efficacy. There is no 'one-size-fits-all' approach to the control or eradication of freshwater pest fish species, and assessments of what method is best will need to be reviewed on a case-by-case basis.

Among the control and eradication options presented above, some of the physical removal methods (e.g., netting, electrofishing, trapping, water removal) and the use of fish toxicants (e.g., rotenone, antimycin, chlorine, lime) are currently considered proven rather than experimental approaches. However, given that it is not uncommon for a combination of control and eradication methods to be deployed simultaneously, resource managers could conceivably consider combinations of the above before deciding how to reduce the impacts of established ornamental fish.

Whatever the approach and method used for pest fish control, resource managers will need to ensure that effective barriers to further spread and public relations programmes to prevent future re-introductions are put into place. There also needs to be stringent risk assessments and communication plans developed for many of these control and eradication techniques. We note that this is something that has been considered as part of the Operational Strategy for Control of Exotic Fishes in Queensland (Mackenzie, 2003).

Regardless of anything covered above, the effectiveness of control and eradication programs can only be quantified if rigorous monitoring programs are put in place that will allow before and after treatment densities of the target species to be determined and/or a reduction in impacts to be measured. This will require the use of pilot studies to determine the adequate number of samples required to detect a change between treatments and controls. The reason why it may be desirable to monitor changes in both populations of the targeted species and those of certain native fish species in association with these programmes is that the goal of resource managers is not only to remove the pest species or reduce their populations to as low a level as possible, but ultimately, to reduce the impacts on native fish and/or the habitats they rely on.

Table 9.1 provides a summary of the relative costs and benefits of the control and eradication strategies discussed above.

Table 9.1: Relative costs and benefits for currently used and viable control methods for pest fish (benefits +, costs -, neither o).

METHOD	APPLICABILITY			DIRECT COSTS			INDIRECT COSTS		
	Eradication possible	Range of species	Range of locations	Labour costs	Equipment & material costs	Frequency of treatment required	Human health risks	Risk to other fauna	Animal welfare issues
Netting, trapping,	NO	+++++	+++++	------	-	-----	o	--	--
Electrofishing	NO	+++++	+++	---	--	---	-	--	-
Line fishing (anglers)	NO	++	+++	-	o	-----	o	-	--
Water abstraction	YES	+++	+++	---	---	-	o	---	---
Rotenone	YES	+++++	+++	----	----	-	--	---	---
Antimycin	YES	+++++	+++	----	-----	-	--	-----	----
Liming & chlorination	YES	+++++	++	--	--	-	-----	-----	-----
Agricultural pesticides	YES	+++	+++	----	---	-	---	-----	-----

10. Future importation status of species on the DEW live import list

10.1 Introduction

Of the 23 species reviewed in this report, ten are currently listed under Part 1 of the DEW live import list (i.e., the schedule entitled 'List of specimens taken to be suitable for live import' - Environment Protection and Biodiversity Act 1999). This means they can be imported to Australia without a permit. Part of the brief for this report was to examine what is currently known about these ten species and to make recommendations to the DEW as to whether they should remain on the Part 1 list (i.e., the do nothing option) or, because of any unacceptable potential environmental cost, be shifted to the Part 2 list of the schedule (i.e., species which do require an import permit from the DEW). A third option would be to remove the species from both lists on the basis that no further stocks should be imported.

The ten species on the Part 1 list are shown in Table 10.1. There are no aquarium species currently on the Part 2 list. There are also many species of aquarium fish in Australia that do not appear on the live import list. Some of these species have been proposed for listing under the Noxious Species and Grey Species list being established by the Department of Agriculture Fisheries and Forestry (DAFF 2005). Confirmation of species on the noxious species list would presumably result in a ban on future importation.

In this chapter of the report, we draw upon the results of the species reviews and knowledge of the known distributions of these species in Australia to assess their relative potential for creating widespread impacts. This 'potential ecological cost' is then contrasted with the relative value of the species to the aquarium industry and the consequences of any restriction on imports to determine whether their importation status should be changed. It should be noted that listing on the live import list is independent of any listing under the Quarantine Act. The latter is designed to reduce risks of introducing pathogens and diseases to Australia.

10.2 Approach taken

An exotic species becomes a nuisance to society when it causes an unacceptable ecological impact that requires a management action to ameliorate it, or would require such action if it were technically possible to provide it at an acceptable cost. However, nuisance species only become pests when they spread widely and such management requirements escalate. To assess the potential risk of an exotic aquarium species becoming a pest fish species in Australian waters we examined firstly the risk that a species will cause a decline in native fauna, especially native fish, and secondly the extent to which it might be expected to spread (i.e., its potential invasiveness).

Table 10.1: Current listing of the 23 aquarium species in relation to their importation status (no aquarium species are currently listed under part 2 of the schedule of fish allowed to be imported). Proposed noxious and grey list species as in Tilzey (2005).

Common name	Scientific name	Current listing		
		Live import list-Part 1 species	Proposed noxious species	Proposed grey list species
Family Cichlidae				
Hybrid cichlid	*Labeotropheus/Pseudotropheus*			
Jewel cichlid	*Hemichromis bimaculatus*			
Victoria Burton's haplochromis	*Haplochromis burtoni*			
Black mangrove cichlid	*Tilapia mariae*		yes	
Redbelly tilapia	*Tilapia zillii*		yes	
Mozambique tilapia	*Oreochromis mossambicus*		yes	
Oscar	*Astronotus ocellatus*	yes		
Three-spot cichlid	*Cichlasoma trimaculatum*			
Jack Dempsey	*Cichlasoma octofasciatum*			
Red devil	*Amphilophus labiatus*			yes
Midas cichlid	*Amphilophus citrinellus*			yes
Convict cichlid	*Archocentrus nigrofasciatus*			
Blue acara	*Aequidens pulcher*	yes		
Family Poeciliidae				
Green swordtail	*Xiphophorus hellerii*	yes		
Platy	*Xiphophorus maculatus*	yes		
Sailfin molly	*Poecilia latipinna*	yes		
Guppy	*Poecilia reticulata*	yes		
Caudo	*Phalloceros caudimaculatus*			
Family Osphronemidae				
Three-spot gourami	*Trichogaster trichopterus*	yes		
Family Cobitidae				
Oriental weatherloach	*Misgurnus anguillicaudatus*		yes	
Family Cyprinidae				
Goldfish	*Carassius auratus*	yes		
Rosy barb	*Puntius conchonius*	yes		
White cloud mountain minnow	*Tanichthys albonubes*	yes		

The former assessment requires some knowledge of fish size, habitats (e.g., river, stream, lake) and feeding ecology (e.g., piscivory, omnivory, herbivory) and whether it displays aggression to other fish species. This assessment also includes any observations of the outcome of interactions with native fauna either in other countries or in Australia. The information for this assessment is taken from chapter 4 of this report.

The latter assessment requires an appraisal of the species capacity to spread unaided once established in the wild. This too requires some assessment of habitats that can be occupied (e.g., still versus flowing waters, fresh versus brackish waters) and it involves consideration of its likely tolerances of water temperature, salinity and oxygen as these have a bearing on its potential geographic range. However, invasiveness also needs to account for spread by vectors such as anglers, aquarists, birds etc. The information for this assessment is also taken from chapter 4.

Removal of a species from the live import list would make little sense if it was already widespread in the wild, especially if its continued importation was of high value to the aquarium industry. Conversely, removal of a species from the list should be seriously contemplated if its risk of becoming a pest is high, its value to the industry is low, and a reduction in its importation would reduce the risk of it becoming more widespread (i.e., propagule pressure is reduced such that the risk of spread and establishment declines).The assessment of whether a species should stay on the live import Part 1 list or not therefore needs to contrast the potential loss to the industry of removing it from this list against the loss to society if it proves to be a pest species and subsequently results in widespread, irreversible ecological damage. To assess the overall cost/benefit of removing it from Part 1 of the live import list (i.e., of reducing its rate of importation) we took into account the number of fish imported by the industry and the overall value of these (see data in chapter 7). We also examined the extent to which a curb on imports might restrict its current distribution within the wild (chapter 2) and whether it could be artificially bred in large numbers in Australia should imports be restricted.

These factors are weighed for each species below and our recommendations are based on a judgement of the overall cost-benefit of retaining each species on the Part 1 list.

10.3 Species assessments

Oscar (*Astronotus ocellatus*): Under the right conditions, oscars can grow to a large size (40 cm) and they have been shown to be capable of feeding on other small fish as well as on invertebrates. The males display aggressive behaviour to other fish during spawning and the species is known to 'burrow' into the substrate (probably during nest preparation). These attributes collectively indicate a relatively high behavioural potential for impact on native fish and invertebrates.

This species also has a high propensity to spread. It has an oval-shaped, laterally compressed body form and is reported to occur mainly in slow-flowing waters such as occur in the lower regions of rivers and in lakes and reservoirs. Its temperature tolerances indicate a broad potential geographic range but this can be expected mainly in the warmest and hence northernmost regions of Australia. It can inhabit degraded waters such as swamps, ponds, canals and ditches where oxygen levels can be low at times. It is regarded as a good food fish and a sports fish in some parts of the world, so can be expected to be spread by humans. This, together with its wide 'potential' geographic distribution and propensity to cause an impact mean that it has a high risk of becoming a pest species.

Even though the volume of oscars imported to Australia is relative low (in the order of tens of thousands) the overall value of importations to the aquarium industry is rated as high. Given its very limited known distribution in Australia at present (i.e., Ross River and Cairns), but its high potential for both impact and spread by human vectors, its future distribution needs to be more tightly controlled and in this respect controls over importation would be useful. It can be artificially bred in warm-water aquaria so its transfer from the Part 1 to Part 2 list of the live import schedule should be seriously considered so long as this will restrict rather than promote its spread in the wild. Its long-term future importation status in Australia will depend on clarification of its potential to create impacts.

Blue acara (*Aequidens pulcher*): The blue acara is a small-sized fish (maximum size 16 cm) and it is reported to be an omnivore, but is also capable of feeding on small fish. It too is a bottom disturber. Like the oscar it has an oval-shaped, laterally compressed body form but it is more elongate and hence tends to occur in faster flowing waters as well as in still-water environments such as ponds and canals. It occurs over a wide range of temperatures but because of its ability to occupy flowing water habitats can be expected to have a wider potential distribution in northern Australia than the oscar. Its propensity for spread by human vectors is potentially lower because of its lower utility.

Currently the number of imports is relatively low (e.g., low tens of thousands) and its overall value is not as high as the oscar. Given this and its current limited distribution in the wild (i.e., heated water from the Hazelwood Power station near Melbourne, Brisbane creeks and a dam) it should also be considered for transfer from Part1 to Part 2 of the live import list schedule and require a permit before importation. It is readily bred in captivity.

Green swordtail (*Xiphophorus hellerii*): The green swordtail is a small-sized fish (maximum size 14 cm) and it has been shown to feed on small native fish in Australia. Although it is generally peaceful in aquaria, it can be aggressive towards other fish and has been associated with the decline of native fish in several Australian studies so is clearly capable of impacting on the native freshwater fauna of Australia.

It has an elongate, cylindrical body form and occupies more rapidly flowing streams and rivers than the blue acara, but also occurs in still water environments. It tolerates a similar range of water temperatures to the blue acara, but can cope with degraded environments where oxygen levels are often low as well as slightly brackish water environments. It is therefore likely to have a similar distributional range to the oscar, but can be expected to occupy a much wider range of habitats within this. For example swordtails have become established in several, small Brisbane streams that are likely to be too cold for the Oscar but which contrain gambusia (pers comm.., Brett Herbert). Although it is used for genetics research in scientific laboratories, it has little other utility than in the aquarium industry and so its spread in the wild is likely to slow and related mainly to releases by aquarium fish adherents. Nevertheless, it already occurs widely if sporadically along the Queensland coast and this probably reflects its greater use in aquariums.

It is of high value to the industry primarily because of the large number of imports (high hundreds of thousands). Despite this high value, its potential for impact across a wide swathe of Australia is high. Given its current limited distribution in Queensland, it too should be considered for transfer from Part 1 to Part 2 of the live import schedule.

Platy (*Xiphophorus maculatus*): The platy is a much smaller fish (maximum size 6 cm) than the swordtail or blue acara and unlike these species is not reported to prey on other fish or to exhibit aggressive behaviour to other species of fish. However, it does eat small invertebrates and so may compete with other fish for food if population densities become very high. This could result in the exclusion of other fish from habitats frequented by platys and hence a reduction in the distribution of the native fauna. Although there are reports warning of impacts by platys in other countries, we could find no data to either confirm or deny these. At present, the platy is widely distributed in many waters in coastal Queensland and so it should be possible to test such concerns about its potential impact.

This species is of high value to the industry and it has a wide geographic distribution along the Queensland coast. A curb on imports would have little benefit in terms of restricting its future spread in this State but would adversely affect the industry. For this reason, removal of this species from the Part 1 list would be of little use in restricting its spread within this State. Should studies of field populations in Queensland indicate that impacts are occurring, the importation status of this species would then need to be reviewed, particularly as this applies to the potential spread of platy within the Northern territory and Western Australia.

Sailfin molly (*Poecilia latipinna*): The sailfin molly can grow up to 15 cm in length and is adapted to occupy still or slow-flowing shallow waters. Although there are few studies of its diet in the wild, it appears to be mainly herbivorous and to feed on algal material. It is also non-aggressive and so any impacts on native fauna can be expected to be mainly indirect (i.e., through changes mainly in food webs). Concerns about its potential impact are therefore low but with the caveat that there are few studies of its feeding in the wild to confirm its herbivorous nature.

Its risk of spread is somewhat higher than that for the platy because wild fish are reported to have larger 'sailfins' and this may encourage its release into the wild by some aquarists to create harvestable populations. It is also tolerant of a broad salinity range which increases the risk of it establishing in inland brackish-water lakes. Nevertheless, it is only reported from a few locations along the Queensland coast so far and is much less widespread than the platy.

At present, it is a highly valued species with imports ranging in the high hundreds of thousands and, apart from the fact that it is currently known from few locations in the wild (i.e., its distribution in the wild is probably very limited), there is no good reason to justify its removal from Part 1 of the live import list.

Guppy (*Poecilia reticulata*): The guppy is, like the platy, a very small fish (maximum size 6 cm) and has not been reported to prey on other fish (apart from its fry). It is omnivorous and is known to be an egg eater, so could affect native fish that spawn on the substrate in shallow waters. There are several reports of impacts in other countries but as yet there is no evidence for these.

Like the platy it is already widespread, occurring in coastal waters along the coast from northern Queensland to New South Wales. It prefers warm waters but has a wide tolerance of water temperature and copes with a range of salinities. Its potential geographic distribution is therefore large, but within this it would tend to occupy still or slow-flowing waters rather than flowing water habitats.

Its widespread distribution in coastal Queensland and New South Wales provides plenty of scope to determine its impact on other fish in the wild in Australian waters and means that there is little point in curbing imports to restrict its spread here, especially as it has high value to the industry. As with the platy, removing this species from the Part 1 of the live import list is not warranted at present. However, should studies of field populations in Queensland indicate that impacts are occurring, the importation status of this species may need to be reviewed to restrict its future spread within the Northern Territory and Western Australia.

Three spot gourami (*Trichogaster trichopterus*): The three spot gourami is a medium sized fish (maximum length 15 cm) and is omnivorous. There are no reports of it preying on other fish or displaying aggression and no reports of it affecting native fish in other countries where it has been introduced. This may be taken to indicate a lack of impact potential, but there are few studies of this species in the wild and so it is characterised by a distinct lack of information.

It is found mainly in still or slow-flowing freshwater habitats and can tolerate low oxygen levels. There are no anthropogenic factors known to enhance its risk of spread and it currently has a very restricted known distribution within Australia. However, in its natural habitat, it is reported to undertake seasonal migrations from standing waters to flood plains and back. This migratory behaviour would enhance its spread within a river system once it is established there.

This species has a high value to the aquarium industry because of the high level of imports and a curb on importation would harm the industry. At present there is too little known about it to warrant a recommendation on its importation status.

Goldfish (*Carassius auratus*): The goldfish is a ubiquitous species found in many parts of the world and is not known to directly affect other fish species. It is primarily a detritivore. There are some concerns about high densities of this species increasing turbidity levels and promoting blue-green algal blooms in shallow, eutrophic waters. There are also concerns about its impact on trout cod (*Maccullochella macquariensis*) presumably as a result of habitat modification. These concerns have not been substantiated to date, and if so, are likely to be limited to habitats where population densities become high.

The goldfish now has a very wide south eastern distribution within Australia mainly occurring throughout New South Wales, ACT and Victoria. This geographic distribution is not expected to change greatly and a curb on importation is unlikely to affect its distribution in the wild. The species is of high value to the aquarium industry and therefore there is no good reason to remove it from Part 1 of the live import.

White cloud mountain minnow (*Tanichthys albonubes*): This species is extremely small (maximum size 4 cm) and does not prey on other fish (apart from fry). It is likely to be mainly carnivorous, feeding on small insect larvae and zooplankton in small streams with flowing waters. It is a schooling fish and can be aggressive to other species in aquaria. There is a distinct lack of information on its ecology in the wild.

Although its potential impact on other fish is unknown, its potential to spread is very high because of both its tolerance of low water temperatures and its ability to cope with flowing waters. In this sense its potential geographic distribution could be expected to match and even exceed that of goldfish. Its spread is likely to occur mainly as a consequence of the release of aquarium fish, but its current known distribution is limited to two sites indicating that, compared with some other aquarium fish species, its accidental release is rare, or that its survival after release is low.

Although this species may not have a major impact on native fish, or be spread rapidly, its potential range is wide and it could be argued that a curb on importation is needed to help restrict its further spread until its potential to create an impact or not can be better established. However, this species has a high value to the aquarium industry because of the high level of imports. A curb on importation could harm the industry and if this species can be readily bred from existing stocks within Australia (e.g., because of its low temperature requirements), then a curb on importation may instead encourage more breeding within Australia and this may then escalate its spread.

Rosy barb (*Puntius conchonius*): The rosy barb is a small-sized fish (maximum length 14 cm) and although it is not reported to prey on other fish it is omnivorous and in aquaria has a reputation as a 'spawn robber'. In aquaria rosy barbs are reported to swim near the bottom and can be aggressive unless present in groups. It is capable of tolerating moderate water velocities and is likely to form schools. There are no reports of impacts on native fish in the wild, but this reflects a lack of study as against a lack of impacts.

Its ability to cope with moderate water velocities (e.g., fast-flowing hill streams) and its tolerance or relatively low water temperatures means that like the white cloud mountain minnow and goldfish, it can be expected to have a wide potential geographic distribution in southern Australia if allowed to spread. In this sense, its potential invasiveness is high, but its potential impact is unknown.

It is a high value- high volume species for the aquarium industry and its known wild populations in Australia are currently limited to two locations. A restriction on importation would possibly disadvantage the aquarium industry and lead to more breeding within Australia with a consequent greater risk of spread.

10.4 Recommendations

Our assessment of the cost/benefit of allowing continued importation without permit of all the species reviewed and currently on the Part 1 list of the live import schedule indicates that three species could be considered for transfer to the Part 2 list (i.e., require an import permit). These are the oscar, blue acara, and green swordtail. Transfer to the Part 2 list implies control over importation by the DEW through conditions on importation permits (e.g., more stringent health certification to be applied by AQIS, restriction on where importation is allowed to restrict spread). But the aim and feasibility of such conditions would need to be fully explored on a species-by-species basis and such a transfer would need to be justified by consideration of the wide range of factors involved, including not only the disease risk but also the potential for environmental impacts on native species (see Chapter 4) and the regulatory implications for both DEW and AQIS. The aquarium supply industry would be opposed to such a list transfer because it would not restrict the spread of these species within Australia, and would make importation potentially more difficult. Thus, justification for such a transfer would be required more on the risk of disease importation than on ecological impact grounds.

There is not sufficient justification in terms of ecological cost/benefit to remove the other species from the Part 1 list at present. This aside, more information is clearly required on the effect that platys and guppies may currently be having on the native fauna in Queensland waters. These species are already widespread in freshwaters down the Queensland coastline and so it will be possible to obtain such information. If an impact can be demonstrated, then their future spread to freshwater habitats in the northern territory and Western Australia would need to be prevented and a restriction on importation to these States may be warranted.

There was too little information to make any recommendation on the three-spot gourami. Although information on the potential for impacts was also lacking for the white cloud mountain minnow and rosy barb, the ability of these three species to cope with lower water temperatures means that breeding is likely to be much easier in Australia than for other ornamental fish species.

The recommendations above are based on ecological cost/benefit considerations and not on threats to fish health posed by introduced pathogens (see Chapter 5). When threats to fish health alone are considered, it is apparent that the goldfish, guppy and three spot gourami pose a greater risk than other species of introducing pathogens into the wild that could prove damaging to native fish health as well as to other fish resources such as salmonid fisheries and freshwater aquaculture industries. However, there is little scientific information on the diseases and parasites present in aquarium fish in the wild in Australia to fully assess these risks and, because of a lack of funds, there there is no surveillance and monitoring to determine their prevalence and distribution. In the absence of such funding, there is scope for more public education to reduce the frequency of hobbyists releasing aquarium fish into the wild.

AQIS imposed new measures on the importation of live ornamental fish to Australia in 1999 including official health certification, pre-export quarantine of 2 weeks, and post-arrival quarantine of up to three weeks. The quarantine risk management measures for the importation of ornamental finfish may be reviewed when relevant scientific information becomes available that demonstrates that current risk management measures may not be effective.

11. Overall summary and recommendations

This review of the impacts of ornamental fish species currently established in Australian waters was not intended to provide an analysis of the regulatory structures for exotic fish in Australia nor to provide a comprehensive set of recommendations for the management of such species. However, it is inevitable that recommendations will arise from a review of impacts. The following recommendations therefore complement those provided by individual States (e.g., DPIQ 2001) as well as those developed for application throughout Australia (e.g., Koehn and MacKenzie 2004; DAFF 2005).

11.1 Distribution and spread of aquarium fish

A total of 30 species of exotic fish used as ornamental pets in freshwater aquariums is now known to have populations in the wild within Australia (Kailola 2000; Lintermans 2004; Chapters 1 & 2 of this report). Clearly, only a small proportion of aquarium fish present in Australia have wild populations as the total number of species kept in aquaria is likely to number in the many hundreds. Nevertheless, the addition of 30 exotic fish species to the Australian fish fauna represents a potentially large change that could have major implications for native biodiversity.

Some families of fish appear to be over-represented in the wild populations. In particular, 19 of these 30 species are cichlids. In addition, there are 5 poeciliid species, 4 cyprinids, 1 cobitidid and 1 osphronemid species. However, no species in the families Characidae, Cyprinodontidae or Callichthyidae are known to be present in the wild despite the high number of species within these families that are kept in aquaria. Reasons for this apparent disparity are not yet known.

The establishment of aquarium fish in the wild has accelerated over the past decade in Australia and, if nothing is done, it is inevitable that in the future many more species will become established and those already present will spread. Apart from the historic reasons for exotic fish introductions (e.g., escapees from farms, sports fish enhancement, bait fish, aquatic insect control) society is now less tolerant of ‘killing’ animals and animal ethics considerations will increasingly result in more unwanted aquarium fish being released alive into the wild. This trend is no doubt exacerbated by films such as “Finding Nemo” which popularize and humanize small, colourful fish. Many aquarium fish species are now widely established in many of the warmer, southern States of the USA, particularly Florida. If northern Australian States such as Queensland are to avoid a similar dilution of the native fish fauna by exotic species, then urgent action is required to halt the introduction and spread of more ornamental fish species.

The brief for this investigation required the assessment of 23 of the 30 species known to be present in the wild. In general, most of these 23 species are utilised in tropical (i.e., warm water > 20ºC) aquaria and can be expected to occur mainly in the more northern regions of Australia but three species thrive in colder waters and so are likely to occur mainly in the south.

The mapping of the known locations of wild populations of these 23 species shows their current geographical range and provides an indication of their prevalence (Chapter 2). In addition, we provide maps for the known distributions of another four of the species that now occur in the wild. It should be noted that for many of these 27 species, the known distribution at this time depends to a large extent on sampling coverage and there are many gaps in this. Hence, the maps only record the known presence of wild populations, not where the species is known to be absent, or where its presence/absence is unknown because of a lack of sampling. No distributional data were obtained for 3 of the 30 species reported to occur in the wild and their status therefore needs to be determined.

Some trends in species distributions were apparent and are noted in more detail in Chapter 2. In particular, most species occur in freshwaters along the Queensland coastline between Brisbane and Cairns and the highest concentration of species occurred in the vicinity of Townsville. This may well reflect a much higher level of sampling in these areas, but without accompanying data on the number of sites or length of stream sampled in the various regions or States this cannot be determined. Another trend is that for many species their wild populations occurred close to major population centres. Again this may reflect sampling coverage (close to main centres) as against the release of such fish close to major population centres, but the available data do not permit such a comparison. Nevertheless, it is apparent that aquarium fish species are being increasingly released into the wild in the often mistaken belief that this is a humane way of disposing of unwanted fish and will cause no harm. The overall extent of aquarium fish populations in the wild clearly makes a compelling case for public education programmes to rapidly ‘debunk’ this false paradigm.

Limitations in sampling coverage aside, it is apparent that some species of aquarium fish are both widespread and relatively common (e.g., goldfish) whereas others, while being widespread have highly localised distributions (i.e., only a few wild populations occur, but these occur over a wide latitudinal or longitudinal range). Other species are highly localised and are currently known to be present in only one location.

Mapping provides a useful tool for the management of exotic fish species and where accurate can be used to monitor spread, determine priorities for management and help identify optimal management strategies (e.g., containment versus control). (NB. mapping is also a key tool for the management of native fish species). The mapping exercise undertaken for this investigation revealed an urgent need for coordination and control over the recording of fish species occurrence (or absence) in Australian waters and the need for integration and/or coordination of such efforts between States.

We therefore propose the following recommendations:

1. **Explore the feasibility of setting up a national freshwater fish database through consultation with the Australian Society of Fish Biology (ASFB) and the relevant State and National agencies**.

2. **In the interim, set up a database on the distribution of aquarium fish species in the wild and collate all records of both presence and absence. In conjunction with this, encourage further field surveys to fill in the main gaps in sampling coverage.**

Other recommendations stemming from the mapping exercise include:

3. **Taxonomic studies are implemented to confirm the identification of species over which there is potential for confusion and uncertainty (e.g., hybrid crosses, oriental weatherloach, cichlids) and more particularly, ensure that there are good keys available to all field biologists to aid the identification of all of these aquarium fish species in the wild.**

4. **Confirm the presence in the wild for the three species reported to be present, but for which no geographic data could be obtained (i.e. blue tilapia, redhead cichlid, Sumatra barb).**

5. **Confirm the presence of breeding populations for species currently known from just 1 or 2 locations and assess their risk of spread along with the feasibility of containment and/or eradication.**

6. **Investigate the causes of 'hot spots' for species incursions in northern Queensland and develop targeted public relations campaigns to counter the release of aquarium species in these places, as well as in Queensland and nationwide.**

11.2 Reviews of impact assessments

The review of impact assessment methodologies (Chapter 3) revealed that a wide range of approaches and methods are used, extending from quick (and therefore relatively inexpensive) desk-top risk assessments to a multi-year, triple-bottom-line impact assessments incorporating hypothesis-based field studies of both the nature and mechanism of impact across a range of both ecosystems and geographic regions, together with economic and social analyses of the costs of these impacts. In practice, most assessments are carried out with limited funds and time and are therefore of limited extent and predictive value. The impact assessments carried out for aquarium fish species in Australia are limited to a few species and are also limited in scope. This is most likely because of funding and time constraints. The huge gap between the information actually required for good decision-making and the information available does not reflect a lack of good scientific methodology. It generally reflects the fact that managers rarely have the time and resources to apply to proper impact evaluation. An exception to this occurs for exotic species whose wild populations are very limited in number. In this situation, impact assessment is limited to one site and this necessarily limits generalisation of the results and hence their predictive value.

Where limited resources and cost/benefit considerations occur, a judgement needs to be made as to the level of impact that is acceptable given the type of assessment that is affordable. In the case of aquarium fish species, where such 'proof of impact' may trigger management actions and costs, this would be best achieved by prior agreement among stakeholders and biologists as to what is an acceptable level of proof (i.e., establishing the 'burden of proof'). This is a social approach to this issue, not a scientific one and it increases the risk of 'Type 1 and II errors'. In particular, accepting that there is no significant impact from an exotic species and being wrong can result in an expensive problem for society that is irreversible and everlasting. The converse, accepting that there is an impact and being wrong, may involve a higher economic cost to start with but this cost will not be everlasting! Such logic supports a more precautionary approach.

The literature search and review of impacts for the 23 aquarium species now in the wild in Australia indicated how little is known of the ecology of these species in their natural range, let alone in Australia (Chapter 4). This lack of information severely limits the quality and predictive power of the assessments and indicates that there is an urgent need to fill the main information gaps.

This aside, it is apparent that a number of the species possess the potential to become widespread pests. In particular, species that are relatively large and carnivorous and which are capable of direct impacts on a wide range of native fish through predation are a major concern. These include the oscar, threespot cichlid, Jack Dempsey, red devil tilapia, and Midas cichlid. These species can all be expected to inhabit still or slow-flowing, fresh-water habitats in the warmer more northern regions of Australia. Of these 5 species, the oscar and Jack Dempsey can tolerate low oxygen levels so can also inhabit degraded waters. At present, the oscar is only reported from 2 locations in Australia which means that impact assessments would be limited in scope. However, the red devil is reported from 3 locations, 2 of which are enclosed, still-water environments and it may therefore present greater opportunities for impact assessment.

Smaller carnivorous fish that are able to prey on smaller native fish are also a concern, especially if they are also aggressive towards other fish species. Such species include the convict cichlid, blue acara and green swordtail. These 3 species all tolerate faster water velocities than the species listed above so can be expected to occur in rivers and streams as well as in standing waters. The convict cichlid occurs over a wider temperature range than the other species but the green swordtail, while likely to be confined to warmer waters can also tolerate saline waters and low oxygen habitats. These two species therefore have a greater invasive potential than the blue acara. Of these species, the green swordtail currently has the widest distribution in Australia with populations in the Northern Territory, Queensland, New South Wales and Western Australia. In addition it is a livebearer so can establish from relatively few propagules. It should be a major priority for investigations focussed on impacts.

Species that are not predators of other fish also need to be considered threats as, although impacts on other fish are indirect, they can nevertheless displace native fish species from important habitats through aggression, competition for food, disruption of reproduction and habitat modification. The black mangrove cichlid, redbelly tilapia and Mozambique tilapia fall into this category. They are all relatively large fish (up to 40 cm long), and although they are primarily herbivorous as adults, when abundant they can potentially displace native fish through aggressive behaviour during the spawning season. Similar behavioural traits have been attributed to the jewel cichlid. Although it is a much smaller fish, such traits can also be expected to result in negative interactions with small native fish. Both the redbelly tilapia and Mozambique tilapia have also been reported to be aggressive to other fish and are substrate diggers. They have a wider temperature range than the black mangrove cichlid, can tolerate brackish water conditions and are more likely to be spread because of their aquaculture and sports fish values. The Mozambique tilapia also tolerates low oxygen levels. Given its slightly greater invasive potential and the greater number of wild populations in Australia it is clearly a priority species for research to determine the nature and scope of such indirect impacts on other fish.

The platy, sailfin molly, guppy and caudo are all livebearers that can establish from relatively few propagules and are closely related to gambusia which is a known pest fish. However, the caudo has been reported to displace gambusia and so may present an even greater threat. It tolerates brackish waters and has been used for mosquito control so may well be spread more readily. It is now present in 3 locations in Australia (2 in New South Wales, 1 in Perth) and is also a priority for impact assessment in these locations.

The remaining 5 species (oriental weatherloach, goldfish, rosy barb, sumatra barb and white cloud mountain minnow) all occur naturally in colder waters than the other species and so have a greater potential distribution within the southern regions of Australia than most of the other aquarium fish reviewed. Although there is little known about the oriental weatherloach it has been associated with the decline of small galaxiid fishes in Australia so is also a high priority for research into impacts. Of the remaining species, the rosy barb and white cloud mountain minnow occur in streams as well as in still water environments. The former species is reported to be an egg eater whereas the latter can be aggressive to other species and is a carnivore. These two species also need urgent investigation to determine whether they pose a threat in Australian waters.

Overall, this review of impacts has revealed significant cause for concern over the environmental impact of some ornamental fish species in Australia but there is very little hard evidence to support such concerns and virtually no definitive knowledge of mechanisms. This situation needs to be addressed urgently so that management of priority species is underpinned by good science. This situation led to our seventh recommendation:

7. More comprehensive impact measurement is urgently required for species identified as high priorities based on the type of impact expected, their invasive potential and existing reports of potential effects on other fish. These include the Mozambique tilapia, oscar, three spot cichlid, Jack Dempsy, red devil, Midas cichlid, convict cichlid, blue acara, and green swordtail.

The assessment of the fish health risks associated with the spread of wild populations of aquarium fish (Chapter 5) underlined the importance of this often neglected issue. The presence of wild populations increases the risk that some diseases and parasites associated with aquarium fish will find their way into native fish populations but, at present, this has not been investigated. This chapter also revealed a lack of information on the pathogens and parasites associated with aquarium fish. An even greater gap concerned the parasites and pathogens of native fish. Aquarium species identified as posing a high risk in terms of the spread of diseases included the goldfish, three spot gourami, and all of the poeciliid species. However, the Mozambique tilapia, oriental weatherloach and rosy barb are also a concern and were rated as medium risk species. Several practical ways of filling the knowledge gaps are recommended. They are:

8. Increased surveillance of the parasites and disease agents of aquarium fish traded internationally;

9. Increased surveillance and taxonomic study of the parasites and disease agents of Australian native fishes in the wild; and

10. Increased surveillance, taxonomic and epidemiological study of the parasites and disease agents of introduced fishes.

Practical ways of mitigating disease threats posed by the establishment of aquarium fish species could include:

11. Increased public education to reduce the frequency of hobbyists releasing aquarium fish into the wild; and

12. Providing Biosecurity Australia with relevant new scientific information to support a review of current quarantine risk management measures when this information demonstrates that current risk management measures may not be effective.

The genetic threats of hybridisation, introgression and breakdown of species boundaries associated with the spread of aquarium fish in the wild in Australia are viewed as negligible (Chapter 6). This is because the Australian freshwater fish fauna is highly endemic and it does not include the main fish families represented by the exotic aquarium fish species now in the wild. However, a future risk may be posed by hybridisation among the exotic species now in the wild with the resultant production of genetically different morphs with increased adaptability to the Australian environment. To minimise this risk, attention needs to be focussed on locations in Australia where two or more closely related exotic species co-occur to see whether hybridisation may occur between them and whether any resultant hybrids may pose a risk. Artificial mating of such species could be trialled under laboratory conditions to test the viability of hybrids. The recommendation to deal with this issue is:

13. Monitor sites where two or more closely related aquarium species now occur together to determine whether any hybrids develop. Test for hybridisation potential by cross-breeding these species under laboratory conditions.

11.3 Socio-economic values of aquarium fish in Australia

The economic analysis of the aquarium fish industry (Chapter 7) revealed a number of surprising statistics that collectively describe the industry. These are listed in Table 11.1 below. Because of the lack of information on impacts that wild populations of aquarium fish species in Australia may be having, no economic or social analysis of these costs could be made. However, a comprehensive review of methods variously used to determine the socio-economic costs of environmental impacts resulted in the recommendation of an innovative population dynamic-based methodology. This could be readily adapted to determine the environmental costs of aquarium fish on native fish, however, the use of such models depends on having adequate 'input' data and this will be lacking until there is a much better quantification of the actual impacts that aquarium fish are, or may be, having on other fish.

Table 11.1: Statistics describing the aquarium fish industry in Australia.

Variables describing the industry	Value (A$) or number
Value of ornamental fish trade in Australia	$350 million/annum
Number of pet shops and aquarium shops	1025
Staff employed	6150
Annual turnover	$970 million/annum
Consumer expenditure on aquarium fish	$75-90 million/annum
Value of aquarium fish exported	$1.4 million/annum
Value of aquarium fish imported	$4.2 million/annum
Number (and % of total) of aquarium fish kept in New South Wales	$4.2 million (34%)
Number (and % of total) of aquarium fish kept in Victoria	$3.0 million (35%)
Number (and % of total) of aquarium fish kept in Queensland	$2.2 million (18%)
Number (and % of total) of aquarium fish kept in Western Australia	$1.2 million (10%)
Number (and % of total) of aquarium fish kept in South Australia	$1.0 million (8%)

It was not possible to compare the socio-economic value or ecological impact of aquarium fish with other stressors of the native fauna including other exotic fish species, altered flow regimes, water quality decline, degradation of habitats or barriers to fish movements (Chapter 8). Comparisons have been made between a number of feral animals (e.g., fox, cats, mice, rabbits cane toads) and carp (Chapter 7), but too little is currently known about the impacts of aquarium fish to permit even a subjective assessment. The estimated cost of the effect of carp in Australia was around $16 million. This is for just one species of fish and if only a quarter of the aquarium fish species now present in the wild in Australia proved to be pests, and involved a similar order of costs, then the total figure could be as high as $120 million.

11.4 Management of aquarium fish

The Department of Agriculture, Fisheries and Forests has produced a draft strategic approach to the management of ornamental fish in Australia (DAFF 2005) and this provides a number of recommendations concerning national coordination of the regulatory framework for issues related to ornamental fish. It proposed a noxious species list and a grey list of species and the information in our review provides information relevant to the status of species on these lists and to the potential addition of other species to the lists.

The review of methods for the eradication and control of pest fish species (Chapter 9) revealed that a wide range of methods and combinations of methods are used to reduce populations of pest fish and that there is ongoing research to develop new methods. This review was necessarily limited to control and eradication methods and so did not deal with the issue of containment. For many species with limited distributions, eradication may not be feasible and containment and minimisation of their risk of spread may be the most viable option. This is likely to be particularly important for populations that are located high in catchments as flooding and subsequent downstream movement of larvae and juveniles can be expected to spread such populations naturally. One of the strategies for containment may be an on-going reduction in abundance to reduce the number of propagules available for spread. However, in most cases, containment today involves public education and monitoring of nearby waters to detect new incursions.

Only some of the control and eradication methods reviewed in Chapter 9 are applicable to aquarium fish. In particular water removal by pumping, and chemical treatment using rotenone or lime, have been used to successfully eradicate small populations of some aquarium fish species in Australia. The review indicated that there is no standard treatment and that the best options needs to be determined for each population within each ecosystem. The review also indicated that control or eradication cannot be carried out in isolation and that some methods will remove native species along with the pest fish. Therefore management of wild populations of aquarium fish species may in some circumstances require public consultation and the preparation of plans that place control within a wider framework. This may need to incorporate cost/benefit studies as well as impact assessments for the control method and will generally require monitoring to determine the effectiveness of control. Public education programmes are also necessary to prevent re-introduction. The draft report on a strategic approach to the management of ornamental fish in Australia (DAFF 2005) recommended that stakeholders agree on control mechanisms to be used for noxious fish. This will be required to resolve issues concerning the safe use of piscicides and potential damage to native species.

The review of the importation status of the 10 species currently on the Part 1 list of the live import list (i.e., require no permit to import) recommended the transfer of 3 species to the Part 2 list (i.e., require an import permit) on the basis of their overall potential cost/benefit (Chapter 10). These species are the oscar, blue acara and green swordtail. There are other species where there is concern over impact but too little is known on the ecological impact (or not) to recommend transfer from the Part 1 to Part 2 list. Several species that are already widespread in Queensland may be having an impact and if proof of impact was obtained then removal from the Part 1 list may help restrict their spread to other States. Conversely, the absence of significant impact would cement their place on the Part 1 list.

Transfer of species from Part 1 to Part 2 of the DEW live import list may prevent introductions from overseas to States where these species are currently absent from the wild. However, it would not affect interstate transfers. Because of this, transfer of these species to the Part 2 list would be opposed by the industry because it would be impractical and not serve the intended purpose of restricting spread within Australia. Self-regulation by the industry coupled with public education at the point-of-sale to the public may prove a more practical and effective way of restricting spread into the wild. Other forms of self-regulation by the industry may prove equally viable therefore means of restricting species spread within Australia through industry self-regulation need to be explored further.

An analysis of the disease and pathogenic risk posed by all 23 species recommended that the goldfish, three spot gourami and all poeciliid species be removed from the live import list and that these species be propagated by breeding in Australia.

14. On the basis of their environmental cost/benefit, the future spread of oscar, blue acara and green swordtail within Australia needs to be restricted. The options for accomplishing this need to be identified and evaluated in consultation with the aquarium industry.

15. In terms of the risk of importing and spreading fish diseases, the Commonwealth obtain and consider further advice on prohibiting the import of the goldfish, the three spot gourami and all poeciliid species, especially the guppy.

11.5 Overview

This report has mapped the distribution, as it is currently known, of 27 species of aquarium fish known to have established wild populations in Australia and it has reviewed information on the ecological impacts of 23 species. In addition to ecological impacts, the risks posed by the introduction and spread of pathogens and by the introduction of alien genomes are also reviewed and discussed. It also presents an economic evaluation of the aquarium fish industry and an initial appraisal of the overall effects of aquarium fish in the context of other stressors of freshwater ecosystems. Although the evidence for negative environmental impacts is currently very limited, it is nevertheless clear that there are major causes for concern with some species. These concerns underpin an urgent need to obtain more information on the type and scope of impacts posed by the various species together with the socio-economic costs of these impacts so that management frameworks to address the need for control can be implemented.

In retrospect, it is perhaps fortunate that so few of the potential hundreds of aquarium fish species that could have been released into the wild have established so far. But it is also apparent that more releases will continue in the future unless action is taken to counter these. This prospect needs urgent attention because the potential environmental problems posed by even a few of the 27 species currently known to be established, let alone additional species, could be hugely significant and much greater than for other common pets such as dogs, cats and birds. Experience has already shown that some native fish species inhabiting the cooler, colder regions of Australia have been negatively affected by the introduction of sports fish such as trout and perch. Now, aquarium fish collectively pose a similar if not greater threat to the native fauna in the warmer, more northern and inland regions of Australia. Because of the greater number of species now present, the chances of at least one of these having a negative effect on the native fauna is high and this prospect can only increase as the number of wild releases rises.

This potential problem requires the cooperation of the DEW and the aquarium industry in agreeing on an acceptable way of dealing with these issues. To this end, this report makes a number of recommendations that address many of the main issues. It is hoped that these will collectively form a platform from which progress can be made to resolve firstly the problem of on-going releases and secondly, the impact that existing wild populations may be having. Although progress will clearly involve a widespread change in the public attitude to the release of exotic fish, it will also involve the formation of a sophisticated and nationally coordinated strategy to evaluate the various risks and to deal with issues at both a federal, state and local level. In this respect it is hoped that both the DEW and the aquarium industry can play a leading role in developing initiatives for this.

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14. Appendices

14.1 List of contact who were sent the first questionnaire.

Organisation Type	State	Organisation	Contact Name	Expertise
Research	ACT	AFFA	Mary Bomford	exotic species
Research	ACT	Wildlife Research & Monitoring	Mark Lintermans	exotic fish introduction
Research	ACT	Wildlife Research & Monitoring	Brendan Ebner	native fish (vulnerability to exotics)
Government	ACT	Biosecurity Australia	Warren Vant	Contact officer for the Policy Review On The Importation Of Freshwater Ornamental Finfish: Risks Associated With Iridoviruses
Research	NSW	Southern Cross University / ASFB	Andy Moore	fish genetics / exotic fish species rep
Government	NSW	NSW Fisheries (name change)	Michael Holloway	NSW Contact Person regarding control of exotic Pest Fish
Government	NT	Northern Territory Seafood Council		Referred to in the NT aquarium fishery status report 2003 is the peak body representing the licenses of the Aquarium Fishery. The NTAA operates under the NT Seafood Council (NTSC). Member are drawn from the Aquarium Fishery and from the NTSC
Government	NT + WA	DPIF&M	Andria Marshall	Program Coordinator- Aquatic Pest Management
Government	NT	DPIF&M	Helen Cribb	Research Officer
		DPIF&M	Alex Beatty	Technical Officer
Research	Qld	Griffith University	Angela Arthington	Ecology of exotic fish risk assessment (Tilapia, Poecillidae)
Research	Qld	Griffith University	Mark Kennard	exotic species and links with disturbance
Research	Qld + NT	JCU	Damien Burrows	fish sampling in FNQ and knowledge of study in the Burdekin. Just started a major exotic pest sampling program in the NT and is keen to help us out and hear about our study.
Government	Qld	QDPIF	Amanda Dimmock	Qld Contact Person regarding control of exotic Pest Fish
Research	Qld	JCU	Alan Webb	Exotic fish ecology (particularly Tilapia and Red Devils)
Catchment Management Groups	Qld	Mary River Catchment Committee	Dale Watson	knowledge on exotics in the Mary River
Catchment Management Groups	Qld	Mary River Catchment Committee	Brad Wedock	knowledge on exotics in the Mary River
Government	Qld	Queensland Institute of Medical Research	Tim Hurst	native species for mosquito control in SEQ (translocations for residential use)
Aquarium fish stakeholder group	Qld	ANGFA	Bruce Hansen, Jeff Gunston	native aquarium fish hobbyist group
Government	SA	PIRSA	John Gilliliand	Marine Invasive Species (but covers freshwater)
Government	SA	PIRSA	Helen Croft	Compliance officer for introduced pests
Government	Tasmania	Inland Fisheries Service	Jean Jackson	Control of fish pest species
Research	Tasmania	University of Tasmania	Peter Davies	fish biologist / salmonids / fish database for Tas
Research	Tasmania	University of Tasmania	Scott Hardie	just finished PhD
Research	Tasmania	University of Tasmania	Rick Stuart-Smith	just finishing masters, plans to explore fish faunas in the Pacific
Research	Vic	Arthur Ryllah	Tarmo Raadik	exotic fish introduction
Research	Vic	Arthur Ryllah	John Koehn	exotic fish introduction
Research	Vic	Primary Industries Research Victoria	Wayne Fulton	fish biologist / risk assessment for salmonids
Commercial	Vic	Lloyd Environmental Pty Ltd	Lance Lloyd	Translocation Evaluation Panel member in Vic
Research	WA		Brad Pusey	exotic species and links with exotics and disturbance
Research	WA	Murdoch Uni	David Morgan	Introduced species in WA
Research	Fiji	Marine Studies Program, The University of the South Pacific	Patricia Kailola	exotic fish introduction

14.2 First questionnaire

We Need Your Help in understanding the current distribution of ornamental fish that have established populations in Australia's waterways.

NIWA Australia has been commissioned by the Department of Environment & Heritage (DEH) to lead an overview of the risks associated with ***established*** ornamental fish in Australia's waterways. This is a nation-wide study that will examine the ecological and socio-economic impacts associated with such species, the suitability of various control and eradication options and, which knowledge gaps require priority attention. For more details about this project, please contact Damian McRae, DEH's project officer for this study (Email: Damian.McRae@deh.gov.au; Ph: 02 6274 2524; Fax: 02 6274 1332). In order to complete our study, we require your assistance in identifying where introductions have occurred (including habitat information where possible) and how far these species have spread so far. We would be grateful if you could complete the following short questionnaire and return it to Anthony Moore at the earliest possible convenience (before 30/11/05). Anthony's contact details are:

Anthony Moore
Graduate Research College
Southern Cross University
PO Box 157 Lismore
NSW 2480
Ph: 02 66 269 437
M: 04 2826 5720
Email: amoore@scu.edu.au

Thank you in advance for your assistance.

Dr Jamie Corfield
Environmental Scientist
NIWA Australia Pty Ltd
j.corfield@niwa.com.au
www.niwa.com.au

NIWA Australia

Your name		State	

Common name of fish	Scientific name of fish	Tick if you know of any wild populations of these in your State
Red devil	*Amphilophus labiatus*	
Midas cichlid	*Amphilophus citrinellus*	
Oscar	*Astronotus ocellatus*	
Convict cichlid	*Archocentrus nigrofasciatus*	
Black mangrove cichlid	*Tilapia mariae*	
Redbelly tilapia	*Tilapia zillii*	
Three-spot cichlid	*Cichlasoma trimaculatum*	
Victoria Burton's haplochromis	*Haplochromis burtoni*	
Jewel cichlid	*Hemichromis bimaculatus*	
Mozambique tilapia	*Oreochromis mossambicus*	
Blue acara	*Aequidens pulcher*	
Jack Dempsey	*Cichlasoma octofasciatum*	
Hybrid cichlid	*Labeotropheus/Pseudotropheus cross?*	
Three-spot gourami	*Trichogaster trichopterus*	
Oriental weatherloach	*Misgurnus anguillicaudatus*	
Goldfish	*Carassius auratus*	
White cloud mountain minnow	*Tanichthys albonubes*	
One-spot livebearer	*Phalloceros caudimaculatus*	
Sailfin molly	*Poecilia latipinna*	
Guppy	*Poecilia reticulata*	
Green swordtail	*Xiphophorus hellerii*	
Platy	*Xiphophorus maculatus*	
Rosy Barb	*Barbus (Puntius) conchonius*	

Please list the contact details of anyone else you think might know of wild populations of these fish in your State.

Can we contact you again to get more detailed information (**Y/N**)?

If Yes, Anthony or myself will contact you by phone.

14.3 Follow-up questionnaire

Phone/email follow up to Questionnaire

THE KEY INFORMATION WE NEED IS:

1. Locations (GPS, or map coordinates, or sufficient info for us to get a map reference (e.g. large pond about 12 km north of Geelong, on the Watt's farm.).

2. Information on habitat (type of water –lake, pond, wetland, stream, billabong, river, maybe habitat type in running waters (e.g. pool, run riffle etc.), max. water depth, location of fish (edge/ middle), capture method/place, substrate composition (silt, sand, rock etc) , presence of plants and other cover, salinity if known, water clarity, other fish present etc.). This could be a nearly endless list. Better to keep it reasonably succinct and general to start with. Can always go back to source later if some key question not asked. In many cases knowledge of environment may be very general.

3. Evidence for the citing (who made observation, what was observed/measured, who identified species, were juveniles seen, when was observation made year and month if known).

4. Whether or not any attempts for control or eradication have been made (method, frequency and location details).

5. Whether population or distribution monitoring is taking place (method, frequency and location details).