6. Impacts associated with genetic changes

6.1 Introduction

The genetic threats posed to native fauna by the introduction of alien ornamental 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 (*Mucculochella 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 alien ornamental 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 alien 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 alien 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 alien 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 ornamental 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 alien 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 alien 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 ornamental 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 ornamental 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 alien ornamental 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 alien ornamental 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 alien ornamental 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 alien ornamental fish has already happened to some degree and has the potential to happen in the future. Hybridisation within alien 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 ornamental fish in Australia

7.1 Economic value of the ornamental industry

Background and approach: The ornamental fish industry in Australia comprises imports of ornamental fish species, breeding (domestic production) of ornamental 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 ornamental fish. The value of all of these activities taken together represents the gross value of the ornamental fish industry.

The data available on these aspects of the ornamental 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 ornamental 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 ornamental fish 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 ornamental fish 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 ornamental fish 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 Natural Resource Management Ministerial Council recently valued the ornamental fish trade in Australia at approximately $350 million per annum (NRMMC 2006). This figure included the input of commercial fish breeding facilities, wholesale traders, retail outlets and the hobby industry.

The 2001-02 value of 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 ornamental fish species.

Amongst the ornamental fish bred in Australia is a subset of alien 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 ornamental 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 ornamental fish 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 alien 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 ornamental fish valued at $338,000 (NSW Fisheries 2003), and Western Australia produced 288,000 ornamental 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).

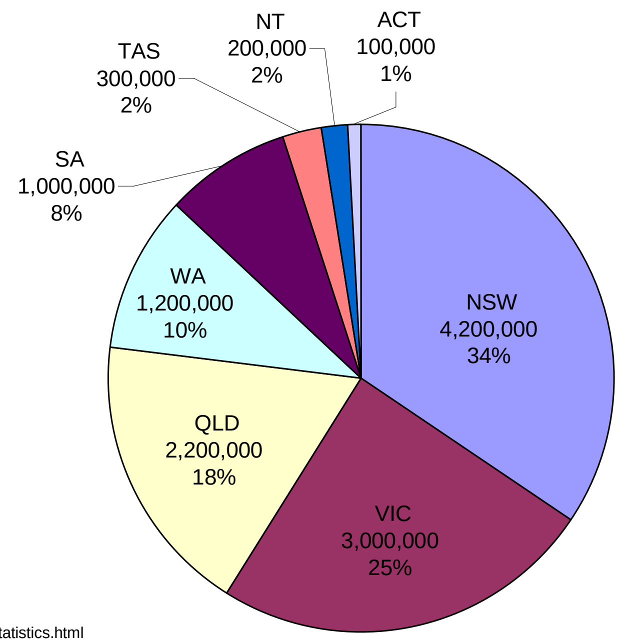
Consumer expenditure and pet fish population: Consumer expenditure on purchasing fish and the various goods and services relating to pet fish are between $75 and 90 million per annum (PIJAC communication).

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 ornamental 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

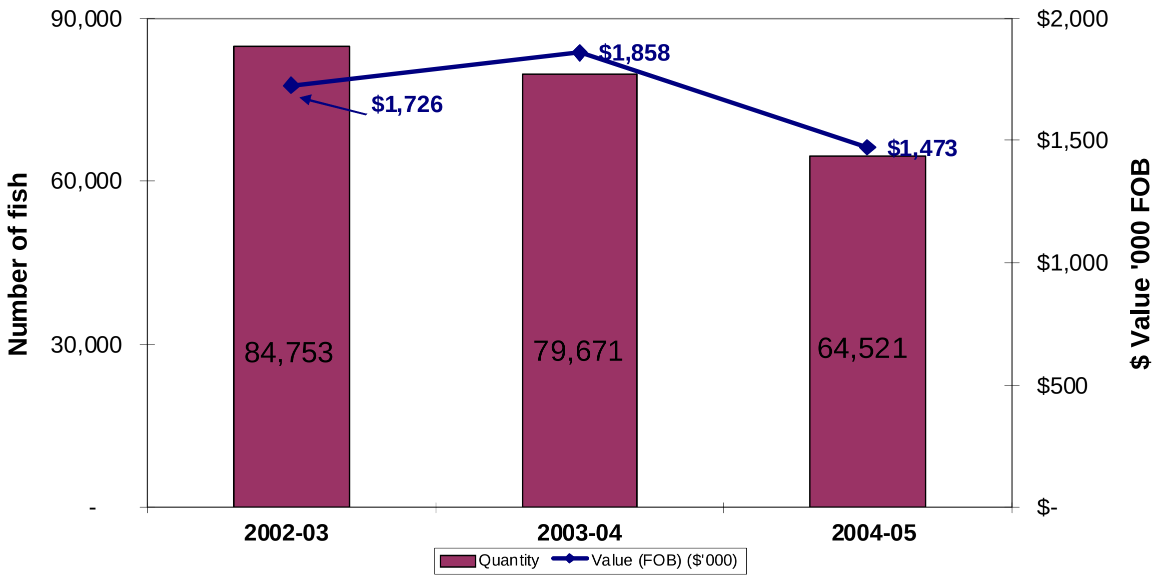
Figure 7.1: Distribution of pet fish ownership in Australia 2002.

Table 7.2: Aquarium retail sector 1999 and 2006.

	1999						2006					
	(A)	**(B)**	**(C)**	**(D)**	**(E)**	**(F)**	**(G)**	**(H)**	**(I)**	**(J)**	**(K)**	**(L)**
	No of stores	**% of total stores**	**Annual Turnover**	**Turnover per store**	**Staff**	**Staff per store**	**No of stores**	**% of total stores**	**Turnover per store**	**Annual Turnover**	**Staff per store**	**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*

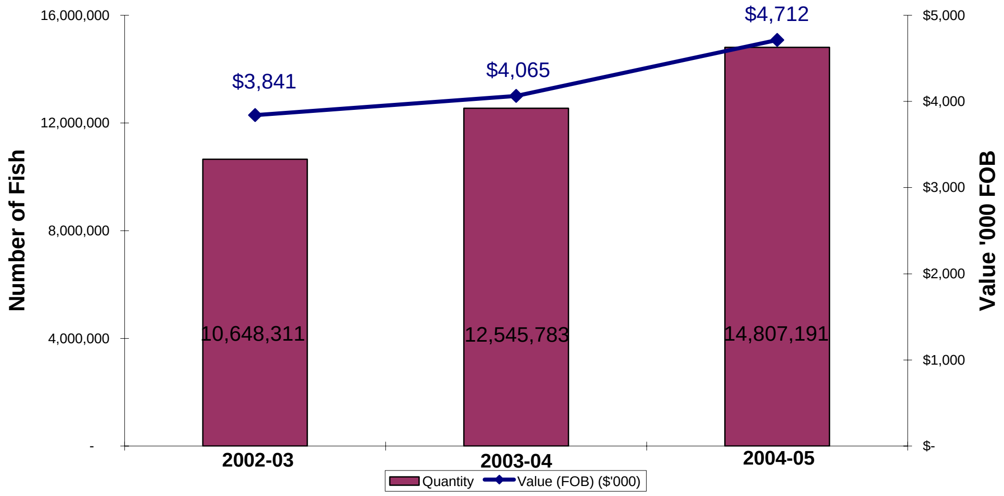
Trade: Over the past decade, the percentage of ornamental fish production (most of which are Australian natives) exported from Australia has undergone a significant decline. In 1995-96, 18.3 per cent of total production was exported overseas, whereas in 2000-01 the figure had dropped to 1.6 percent (ABS 2002;DPI 2002) Further, ABS data indicate that the value and quantity of 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 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 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, are freely traded. These species contribute to the value of the industry and its value would be affected if trade in these alien 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 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) surveyed the impacts of various invasive species. Impacts were classified as economic, environmental or social. Economic cost impacts of aquatic vertebrates, specifically carp, were 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) commented 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 was carp, other species also need to be considered. For example (Agtrans p16) stated: “In addition to carp, there were 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 included:

- 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, they identified no estimates of the economic impacts of introduced freshwater aquatic vertebrates other than those from carp.

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

There was 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 referred to the work of McLeod. In addition, Agtrans outlined impacts attributable to other species:

Other introduced fish also had a negative impact on the environment. These included:

- 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 had been recorded for mosquitofish, swordtails, redfin perch, brown trout, rainbow trout, European carp, goldfish and possibly Oriental weatherloach. There were 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 was still largely unknown, however there was circumstantial evidence of some impacts, as identified in the list above." (Agtrans 2005).

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

"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 is summarised in Table 7.5 and the 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 an ideal candidate 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) concluded 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 ornamental 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 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 management plan for ornamental fish (NRMMC 2006) 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.

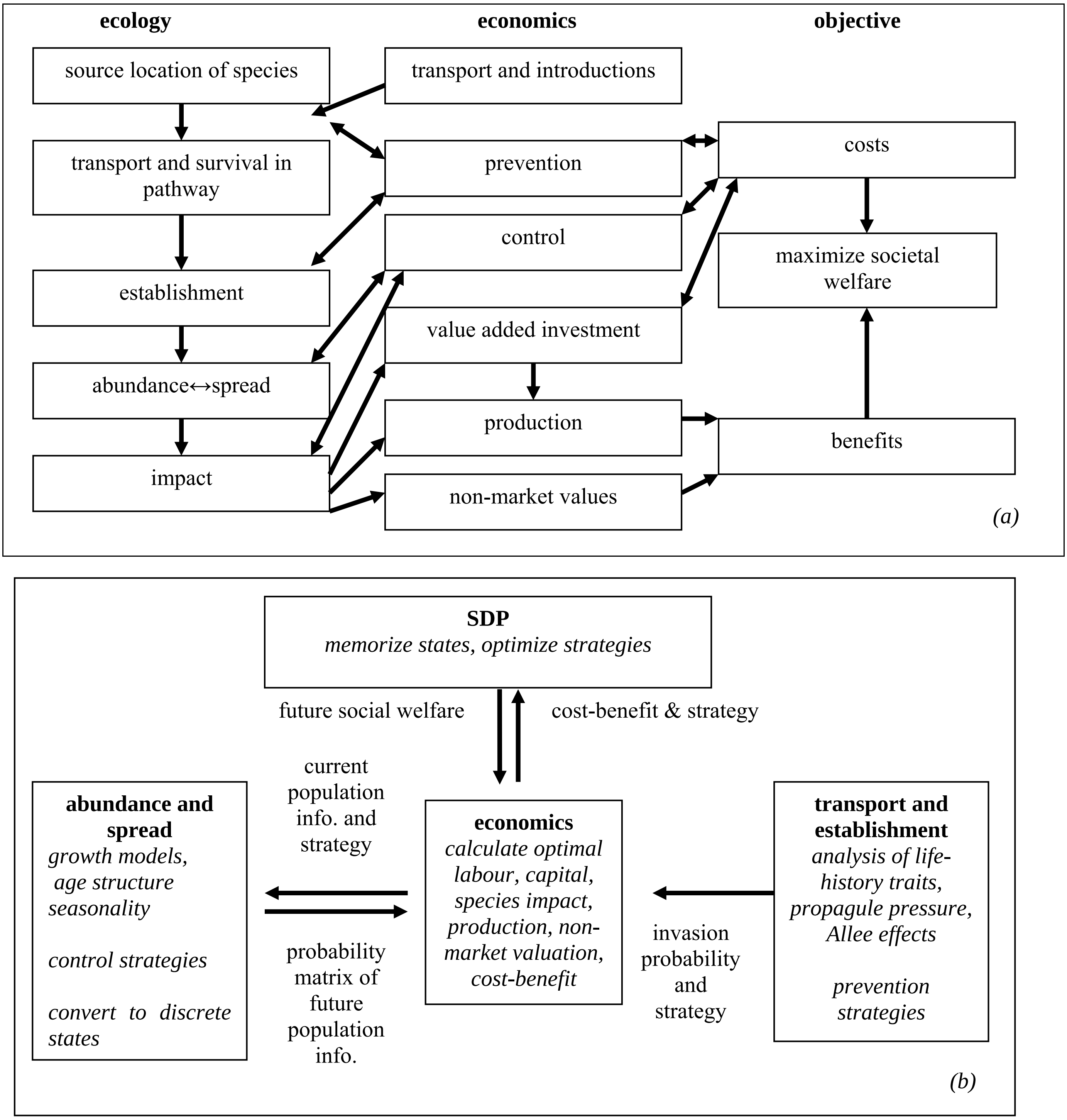


Figure 7.5: Bio-economic framework for invasions: (a) the conceptual approach to the ecological and economic components of a generalized invasion process. Both economic input and ecological states change over time and influence one another. Our goal is to determine the optimal set of strategies that 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 ornamental fish in relation to other stressors

8.1 Introduction

Any impacts of ornamental fish need to be considered alongside those created by other alien 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 alien fish species other than ornamental 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 ornamental fish, principally because a lot more is known about the uses, impacts and management of the non-ornamental fish species. The lack of information on ornamental fish impacts, and the fact that most ornamental fish are currently known from far fewer locations, severely limits any quantitative comparison.

This aside, it might be argued that the impact of ornamental 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 ornamental 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 alien 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 ornamental 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, ornamental fish may too contain the potential for measurable, widespread impacts.

Despite the lack of evidence that ornamental fish are currently impacting on the native fauna, there is enough now known about the behaviour of some ornamental 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 ornamental fish have less of an impact than other alien 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 alien 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 altered flow regimes, degraded water quality, physical habitat removal or modification-in-stream, other alien fish and 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 alien 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.

		Ornamental alien fish	Other alien 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 some 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.

		Ornamental alien fish	Other alien 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, although thermal pollution might affect 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.
		Ornamental alien fish	**Other alien 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 – although 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.

		Ornamental alien fish	Other alien fish	Flow regime	Water quality	Fish habitat	Barrier to fish
	Reduced iconic species	Perceived – for some species.	Perceived – e.g., Murray Cod stocks could be reduced by disease.	Yes – although 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 required.	Perceived - threats to numerous species of galaxiids and several species of pygmy perch.	Yes – although 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.)	Unknown- barriers to Australian Grayling in Victoria have been modified to increase fish passage specifically for this species (Jacques Boubee, NIWA pers. comm.)
		Ornamental alien fish	**Other alien 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 alien 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 alien 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 alien 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 alien 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 alien 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 alien 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 alien 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 Crescent and Sorell in Tasmania, but eradication is proving more difficult and whereas it may be possible in Lake Crescent, it may not succeed in the much larger Lake Sorell (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 Sorell 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 alien 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 alien 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 may not always dispose of

[7] Carp Watch is the only group dedicated to the recording and removal of carp from Australian waterways.

fish in a humane way. However, a greater risk is that anglers targeting alien 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 Tinaroo, 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.), control of tilapia by the Queensland Department of Primary Industry (pers. com. Brett Herbert), caudo control in Bull Creek, Western Australia (Morgan & Beatty 2006a) and goldfish control in the Vasse River, Western Australia (Morgan & Beatty 2006b). 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 waterfall 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 Angourie 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.

As mentioned above, the 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. 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 alien 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 three years later in the Yallourn storage dam in the LaTrobe 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 & Creese 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 alien 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 ornamental 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 alien 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 parasites or pathogens (e.g., fungi, 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 epizootic haematopoietic necrosis (EHN) virus was accidentally introduced and apart from killing large numbers of redfin perch (Langdon & Humphrey 1982) has caused high mortality in some wild fish populations. The introduction of the spring viraemia of carp virus (*Rhabdovirus carpio*) to Australia for control of common carp has been discussed since the 1970's (Crane and Eaton 1996), but this control method has not, to our knowledge, been implemented here because of 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 the common carp is a pest species.

Fishes that are stressed are more likely to be susceptible to the impacts of pathogens. The effectiveness of pathogens will be governed by environmental conditions (such as temperature) and parasites can be expected to 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 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-Millawa 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.

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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.

Both Arthington et al. (1983) and Webb (1994) found that a number of ornamental fish species in northern Queensland waters were thriving in waters where degradation of the habitat had occurred through urban development. Development including, removal 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 alien fish. As a consequence they advocated habitat restoration to change the balance between alien 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 ornamental species (c.f., Arthington et al. 1990)

Pritchard et al. (2004) have also advocated habitat manipulation to restore the balance between native and alien 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 survival of juveniles through lower levels of intra-specific competition. Consequently, 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 Alien Fishes in Queensland (Mackenzie, 2003).

Regardless of anything covered above, the effectiveness of control and eradication programs can be quantified only 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	+++	+++	----	---	-	---	-----	-----