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NIWA Australia

TO THE AUSTRALIAN GOVERNMENT DEPARTMENT OF THE ENVIRONMENT, WATER, HERITAGE AND THE ARTS

Review of the impacts of gambusia, redfin perch, tench, roach, yellowfin goby and streaked goby in Australia

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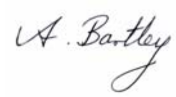


1. Introduction

The introduction and spread of alien (i.e., non-native) species is regarded by many as a major threat to global biodiversity and hence ecological sustainability (Vitousek et al. 1997; Kolar and Lodge 2001; Sakai et al. 2001; Lee 2002; Dudgeon et al. 2006). In particular, studies of fish introductions to freshwater ecosystems in the Northern Hemisphere have shown that some species can reduce native fish populations, degrade aquatic habitats, compromise gene pools, and increase the risk and spread of alien diseases and parasites. As a consequence, the introduction of alien fishes is a major cause of biodiversity decline in freshwater ecosystems (Courtenay 1990; Courtenay and Stauffer, 1990; Courtenay and Moyle, 1992; Fuller et al. 1999; Canonico et al. 2005) and, on a global basis, fish introductions are a prime cause of the extinction of many indigenous fish populations (Sala et al. 2000; Reid et al. 2005).

The Australian continent contains a relatively large number of endemic aquatic fish species (Allen et al. 2002) that form unique component to the biodiversity of its aquatic ecosystems. The wide range of climate types present (e.g., tropical, continental, temperate, island) combine with its diverse geological terrain to provide a large number of fish habitats and these contribute to the relatively high biodiversity of the endemic fish fauna. But these factors also mean that the Australian freshwater fish fauna is vulnerable to the impacts of alien fish species. Leprieur et al. (2008), in their analysis of the main factors associated with alien fish invasions (i.e., human activity, species-rich communities that fill all available niches, the prevalence of suitable habitats), found that human activity, which both modified aquatic environments and increased propagule pressure, was the main driver of alien fish invasions. The presence of suitable habitats and diverse native fish communities was less important, but these factors are clearly needed for both the establishment of alien fish species and for their impacts on endemic biodiversity once an alien species is introduced through human activity. In Australia, land and water use have reduced and modified fish habitat for the endemic species and so created vacant niches for invasion by alien species. There is therefore much potential for impacts of alien fish on endemic biodiversity in Australia, especially in northern and western Australia as human activity there increases. However, this will depend largely on the introduction and spread of alien species through human activity.

Over 40 species of alien freshwater fish already occur in the wild in Australia of which 30 are ornamental species (Corfield et al. 2008). It is therefore no surprise to find that southern Australia is one of six major global hotspots for invasion by alien fish (Leprieur et al. 2008). Some of these fish (e.g., common carp *Cyprinus carpio*) cause environmental damage that has been widely publicised in Australia. However, a number of Australian studies have also expressed concern over the potential impact of other alien fish species on native fish and aquatic habitats (e.g., McKay 1984; 1989; Arthington 1989a; 1991; Clunie et al. 2002; Morgan et al. 2004). If these species were to spread further, they too could create more significant environmental, economic and health hazards. Identification of the species that pose a threat to aquatic habitats and the native fauna of Australia is therefore required as a basis for creating policy to restrict their spread and to manage their wild populations.

As there is no 'generalised invasion theory' that identifies the key properties of potentially harmful alien species, assessments of ecological impacts are required for each species now present in Australia. These need to be based on an expert appraisal of the ecology of each species, the potential for spread, and the likely interactions with native fauna and aquatic habitats. However, such species-specific assessments can only be based on the existing scientific literature, which is likely to be sparse for many species at present. Such assessments will therefore need to include a careful appraisal of what is known about each species and its wild populations and identify the key information gaps that need to be addressed.

Ecological impacts from alien fish can be expensive, if not impossible, to mitigate; especially if this involves the removal of large numbers of fish from certain waterways. In addition, some introduced fish now form the basis for valued recreational fisheries whereas other species may be used in aquaculture or for bio-control. The users of these fish may be opposed to their removal or control in some waters. The assessment of ecological impacts therefore needs to be considered within a broader framework that reflects the value of each species to society and the difficulty of managing it. The legislative and control options available to manage freshwater fish species are also important as they may affect management options. The mitigation of impacts from alien fish in Australia's waterways therefore combines ecological, social, economic and legislative elements. There may be major knowledge gaps in some or all of these elements that need to be filled before effective management options can be agreed by all key stakeholders.

Because of these diverse requirements, the Department of the Environment, Water, Heritage and the Arts (DEWHA) commissioned a series of comprehensive reviews to identify both the current ecological threats posed by alien freshwater fish, and their socio-economic status. The purpose of these reviews is to bring together all the information relevant to the impacts of individual species and to identify gaps in knowledge so that DEWHA can develop and support initiatives to better manage the environmental consequences of these species. To date, three reviews have been commissioned, one for alien salmonids, another for alien ornamental fish in Australia, and a third review to cover the remaining alien fish species, apart from common carp.

This report provides a comprehensive review of these remaining six species, which are listed in Table 1.1.

Table 1.1: Species of alien fish present in the wild in Australia and included in this review.

Scientific name	Common name	Max. total length (cm)	Max. age (yr)	Origin	Typical habitat
Gambusia holbrooki	eastern gambusia	8	1.3	Eastern USA	Still, shallow waters in ponds, lakes, rivers and wetlands
Perca fluviatilis	redfin perch	51	22	Central Europe	Large rivers and lakes
Tinca tinca	tench	84	36	Central Europe	Lakes, ponds and the slow-moving, lower reaches of large rivers
Rutilus rutilus	roach	46	18	Central Europe	Large rivers and lakes
Acanthogobius flavescens	yellowfin goby	30	13	Eastern Russia to northern China	Harbours, estuaries and the lower reaches of rivers
Acentrogobius pflaumi	streaked goby	7	6	Japan to the Philippines	Harbours, estuaries and the lower reaches of rivers

Because of the comprehensive nature of this review and the need to maintain a balanced approach to the issue of whether any of these fish pose a significant threat to native Australian freshwater fauna, a number of experts in various fields contributed to this report. They include Dr David Rowe (National Institute of Water and Atmospheric Research Ltd., NZ) and Dr Anthony Moore (ACT, Australia) who are both experts on alien fish, Annabelle Giorgetti (NZ) who is an economist and director of Enveco Ltd. a company specialising in the provision of advice on environmental economics, Catriona Maclean and Phillipa Grace who are planners with the Engineering and Planning Consultancy Beca Pty Ltd. in Melbourne. Sanjay Wadhwa (NIWA) carried out the GIS-based production of the distribution maps. The project manager and editor of the final document was Dr Jim Cooke (Diffuse Sources Ltd., Wellington, NZ).

Another key factor helping to maintain a balanced approach to this review was the establishment of a peer review panel comprising key stakeholders encompassing the research, fish management and conservation sectors. The review panel was charged with reviewing the report and ensuring that it presents a comprehensive, objective and unbiased viewpoint. Personnel on this panel are:

Science community: Dr David Morgan (Centre for Fish and Fisheries Research, Murdoch University, Perth) is an acknowledged expert on alien fish and is currently researching the threats posed by certain species of alien fish (including redfin perch and eastern gambusia) in Australian waters. He is a member of the Australian Society of Fish Biology’s Alien Species Committee and, as a leading researcher in this field. Dr Morgan is eminently qualified to ensure the review has a strong scientific underpinning.

Fish management: Dr Andrew Sanger, is a fisheries biologist with expertise in native fish conservation and fishery management. He is now Manager of Animal and Plant Regulatory Operations with the Department of Primary Industries in New South Wales (NSWDPI). Among its wide responsibilities, the NSWDPI has management responsibilities for freshwater sports fish including alien and native species as well as biosecurity within New South Wales. Dr Sanger is well placed and qualified to ensure the review is relevant to the management of the fish species addressed.

Conservation: Andreas Glanznig was a former policy analyst in the Federal Environment Department and then the Leader of the Biodiversity Programme with the World Wildlife Fund in Australia. He is now the Chief Operating Officer with the Invasive Animals CRC (IACRC) in Canberra. He has worked over many years to obtain effective control of alien plants and animals in order to reduce threats to Australia’s biodiversity. He provided excellent input to the review of threats posed by ornamental fish species in Australia and has a good overview of the conservation implications of alien fish. He therefore provides a breadth of conservation experience and expertise related to the threats posed to indigenous fauna by alien fish species.

1.1 Study brief

The specific brief for the review from the Department of Environment and Water Resources was:

In relation to the environmental impacts:

1. Provide a summary of the introduction into Australia of roach, tench, redfin perch, eastern gambusia, streaked goby, and yellowfin goby, their current distribution, human utilisation, and biology.

2. Review research findings on the environmental impacts (both positive and negative) of introduced roach, tench, redfin perch, eastern gambusia, streaked goby, and yellowfin goby. This review must:

   2.1. clearly indicate the nature of the environmental impacts at genetic, species and ecosystem levels (both positive and negative), location (where specific Australian locations can not be provided, broader descriptions such as state or bioregion name should be used), and literature reference(s);

   2.2. critically review the research methods, including experimental designs, used to measure and monitor the environmental impacts (both positive and negative) of these introduced fish species in Australia. The strengths and weaknesses of the various experimental designs must be clearly identified;

   2.3. following from 2.2 above, critically review the research based evidence on the environmental impacts (both positive and negative) of these introduced fish species in Australia and assess the strength of that evidence and overall quality of the research;

   2.4. compare and assess the environmental impacts (both positive and negative) of these introduced fish species against other threats (such as the impact of altered environmental flows). This comparison and assessment should be in the form of a general overview, and the use of case studies where the range of other threats is site specific;

   2.5. identify and prioritise gaps in our knowledge about the environmental impacts (both positive and negative) of these introduced fish species in Australia;

   2.6. recommend practical experimental designs for research to fill the identified gaps in our knowledge about the environmental impacts (both positive and negative) of these introduced fish species in Australia.

3. In relation to the social and economic impacts:

   3.1 review the social and economic impacts (both positive and negative) of introduced roach, tench, redfin perch, eastern gambusia, streaked goby, and yellowfin goby in Australia;

   3.2 provide an overview of the economic value of the industries in Australia that are based on these introduced fish species;

   3.3 consider the social and economic impacts (both positive and negative) as they relate to both industry (e.g., recreational fisheries) and conservation management initiatives (e.g., being able to potentially undermine threatened species recovery programs);

3.4 critically reviewing the research methods used to determine the social and economic impacts identified above. The strengths and weaknesses of the various experimental designs must be clearly identified;

3.5 identify and prioritising the gaps in our knowledge about the social and economic impacts of these introduced fish species in Australian;

3.6 recommend practical experimental designs for research to fill the identified gaps in our knowledge about the social and economic impacts of these introduced fish species in Australia;

3.7 In addressing this component of the project, a social impact assessment process will be undertaken using workshops and focus groups in Victoria, New South Wales and Queensland.

4. In relation to the management of these introduced fish species in Australia:

4.1 review and evaluate the current tools, techniques and practices used in relation to the humane capture, handling or destruction of introduced roach, tench, redfin perch, eastern gambusia, streaked goby, and yellowfin goby;

4.2 include a clear description of each of these tools, techniques and practices (including any standard operating procedures and/or codes of practice to guide managers in the use of these tools, techniques and practices);

4.3 focus on the non-target impacts and animal welfare aspects of using these tools, techniques and practices to manage these introduced fish species in Australia;

4.4 review the cost effectiveness of these tools, techniques and practices. This review will include an emphasis on the effectiveness of each option in managing these introduced fish species for the protection of threatened species and ecological communities;

4.5 identify and prioritise the gaps in our knowledge about the humaneness and cost effectiveness of the tools, techniques and practices identified above;

4.6 provide recommendations on areas of future research that will fill these gaps.

5. In relation to the policies and regulations of each Australian jurisdiction that address the issue of introduced roach, tench, redfin perch, eastern gambusia, streaked goby, and yellowfin goby:

5.1 provide a brief summary of the these policies and regulations, and

5.2 discuss the degree to which these policies and regulations are integrated with national policies such as the National policy for the translocation of live aquatic organisms – Issues, principles and guidelines (MCFFA, 1999 (available from: http://affashop.gov.au/product.asp?prodid=12105).

6. Provide a bibliography of all reference material reviewed.

1.2 Report aims and structure

It was acknowledged by DEWHA that not all of the requirements in the study brief could be fulfilled at present because much of the information is lacking. For example, the data needed for adequate ecological and economic assessments was known to be lacking for some species, and it is difficult to compare the impacts of such species against other threats such as environmental flows, without such basic data. Furthermore, much of the information required (e.g., an overview of impact assessment methodologies and management tools to control alien species) is already presented in the review of ornamental fish (Corfield et al. 2008) and so does not need to be duplicated.

With such provisos in mind, this report addresses the brief above by presenting the information obtained within the following chapters:

Chapter 2: Provides an up-to-date map of the 'known' distribution of each species in Australia indicating the river catchments in which they are known to occur as a proxy for the geographic scale of potential impacts.

Chapter 3: Reviews impact assessment methods applicable to the species listed above and required to properly assess impacts. Recommendations on appropriate methods for the species reviewed are included.

Chapter 4: Summarizes the relevant biological information for each species, provides an account of its introduction and reviews studies on its ecological impact both globally and in Australia.

Chapter 5: Provides a review of the potential for genetic impacts such as hybridisation.

Chapter 6: Overviews the social values and potential impacts of the species including the results of the focus workshops in New South Wales, Victoria and South Australia.

Chapter 7: Provides an economic assessment of the species that are of known commercial value in Australia and an evaluation of the impacts of other fish and animals to provide a context for evaluating impacts of the species reviewed.

Chapter 8: Overviews the management methods and tools available to control these species.

Chapter 9: Provides a summary of the legislative framework for the management of the species.

Chapter 10: Summarises the known impacts of these species and presents recommendations for future investigations and initiatives needed to ensure that any ecological impacts of the species are properly addressed and mitigated.

Chapter 11: Provides a listing of all sources for the information presented in the report.

Throughout this report, we use the term 'alien' to identify species new to Australia. In the past, the term 'exotic' has been used to refer to such species but 'exotic' can also include native species that are transferred to a new location and species that are 'unfamiliar' and/or 'unusual'. Alien is a more specific and accurate term, now used widely when referring to a new species introduced to a country from another.

2. Fish distributions

2.1 Introduction

Knowledge of fish distributions is an essential component of environmental impact assessments for alien fish. Maps of species geographical distributions are needed to determine the scale of potential impacts, but they are also required to assist with management. For example, they allow the spread of a species to be monitored, help identify new incursions, assist in the identification of the type of management required at different sites and help determine the priority sites for management.

In practice, good maps of alien fish distributions are rarely available at all the scales required. For example, an overall evaluation of environmental effects of alien fish species within Australia requires a broad geographic scale, encompassing all States, Territories and offshore islands. However, this is too large for the management of individual alien fish populations. River and stream reaches or even catchments and waterbodies occupied by discrete populations of the alien species are the basic units for management of populations and need to be identified at a much finer scale. Clearly, data on species occurrence need to be produced at the finest scale possible (e.g., locations within waterbodies) for management purposes, but they need to be portrayed at a broader national scale to monitor spread. A GIS-based mapping system allows this versatility, hence the data have been collected and collated in a GIS format.

The objective of the species mapping undertaken for this report was to provide a record of each species' 'known' geographical distribution within both the Australian continent and Tasmania that can be presented at a range of scales up to a continental one. Known geographical distribution differs from actual distribution because sampling coverage is very low in Australia and some species may be present in waters not yet sampled. Furthermore, some species may now be absent at sites where they formerly occurred. Correction of such records was beyond the scope of this review, but is clearly needed for the future management of alien species. The 'known' distribution for each species therefore portrays the records available to us as at 2007.

Observations on the location of the alien freshwater fish species in Australia have been collected now for nearly a century in some states, and distribution maps are presented in a number of general publications on Australian freshwater fish (e.g., Cadwallader and Backhouse 1983; McDowall 1996; Clarke et al. 2000; Allen et al. 2002). Most of these publications show fish distributions at a state-wide level and only Allen et al. (2002) provides coverage at a continental scale. However, the maps in Allen et al (2002) do not show the location of individual populations.

The collation of information in historic publications can be expected to indicate the general distribution of each species within Australia and serve as a starting point. However, sampling has increased markedly over the past decade and there are many new records of fish occurrence now present in the scientific literature and especially in the grey literature. In addition, there are now many computerised databases of fish distributions created by both individuals and organisations such as the Australian Museum, the Western Australian Museum Centre for Fish and Fisheries, the New South Wales and Queensland Departments of Primary Industry and the Australia New Guinea Fish Association. The data in these can extend and amplify the historic data. Our aim was therefore to collate as many as possible of the records of fish location for the six species reviewed in this report and to present maps of these data to show the known locations for populations of each species throughout Australia. These data portray the ‘known’ geographic distribution of the species in Australia at this time.

There are a number of inherent limitations in such maps that need to be noted *a priori*. The first is that they do not present records noting where a species has not been found (i.e., absence as against presence). Such data were too difficult to obtain even at a state level. Furthermore, the maps only record where a species has been found because sampling has revealed its presence. They do not record where there has been inadequate sampling. Thus, the maps provide a good picture of where each species is currently known to occur and not its actual current distribution or its potential distribution should it spread. As sampling coverage increases, the gap between the known and actual distributions can be expected to decrease.

A further limitation in these data is the age of the records. Many records were not dated and so it is currently not possible to compare the records over time to determine whether a species is still present at a particular location or not. This is particularly important for alien fish species as some populations may no longer exist, because either they have been eliminated by managers to control their potential spread and ecological impact, or have died out naturally.

These limitations in sampling coverage are important when considering the distribution of species within river networks. For example, records may indicate that a species is present in the middle reaches of a river catchment. However, its absence in the rest of the catchment above and below this point cannot be confirmed if there has been little or no sampling. Larval and juvenile fish are readily transported downstream by flood flows allowing a species to colonise all suitable habitat downstream of a source of adults. However, upstream colonisation is more difficult, especially for fish that are poor swimmers, and these include some of the species reviewed here. Therefore the occurrence of a species in a catchment may well indicate that it is now also present in all suitable habitat further downstream but not necessarily upstream. To encompass this aspect of fish distribution, the distribution maps presented show both the locations of records for each species and the catchments and sub-catchments that they occur within and hence their potential downstream spread through natural dispersal.

2.2 Methods

Data collation and quality control: Data on the distribution of the species reviewed was sought from a wide range of sources. Geographic distributions are portrayed in several general books on Australian fish (e.g., Cadwallader and Backhouse 1983; Llewellyn 1983; McDowall 1996; Allen et al. 2002) and these provided an initial model of the distribution of redfin perch, roach, tench and eastern gambusia in Australia. However, the yellowfin and streaked goby are recent immigrants so no distribution maps for these species are available yet.

The maps that were available reflect current geographic distributions in Australia, but need to be updated because there has been a proliferation of new records over the past decade and in some cases these records will have expanded the known range. Data on the distribution of these species were therefore requested from a wide range of fish biologists, contacts and organisations throughout Australia. These sources are listed in Appendix 1. We initially contacted a key group of people with known expertise and knowledge of alien fish species in each state. Based on their advice, we expanded the list of contacts accordingly. In addition, new locations for the species were retrieved from the recent published, unpublished and internet-based reports that we reviewed for impact assessment purposes. A number of individuals and organisations (e.g., the Australian Museum, Queensland Department of Primary Industry and Fisheries, New South Wales Department of Primary Industries, Murray-Darling Basin Commission, Australia New Guinea Fish Association) also kindly provided us with the data in their databases and, where requested, license agreements were obtained for the use of such information. All data on the two species of coastal goby reviewed were obtained from publications and recent reports on their occurrence in the harbours from which they have been reported. Individuals with some knowledge of fish species in other ports (e.g., Brisbane and Adelaide) were contacted to determine the existence of recent marine surveys that might have revealed the presence of the goby species there, but apart from a single and new record for yellowfin goby near Perth, we are not aware of any such studies in these other areas.

In many cases, the latitudes and longitudes for the records were available. Where they were not (i.e., because only the names of locations or water-body where a species was recorded from were known) the latitude and longitude was determined from the publically accessible, internet-based version of MapConnect. This GIS map of Australia has layers showing, among other features, the place names, roads and all water bodies (perennial and non-perennial) in Australia. It was constructed by Geoscience Australia using Geodata Topographical 250k Series 3 Maps. The mid-points of lakes, reservoirs and other static water bodies were used for determining the coordinates of species records in still-water environments. Where a species was known to be in a stream or river and the exact location of the record was unknown, only the coordinates of the lowest point in the immediate catchment where suitable habitat could occur were recorded. This coordinate selection process was based on the assumption that if the species was found somewhere upstream, it could also be present in suitable habitats downstream at such a site (or would eventually occur there). This process will have reduced the upstream range for some species. In relatively inaccessible areas, the fish record can be expected to correspond closely with the point where a road crosses the river or stream and so provides access for sampling. Accordingly, the coordinates of these sites were determined to mark the location of a record. A number of historic publications and reports (e.g., Mees 1977; Cadwallader and Backhouse 1983; Llewellyn 1983) recorded the location of alien species as a dot on a map. The latitudes and longitudes of these locations were estimated using both MapConnect and GIS based techniques to overlay the distributions on scale-corrected maps so that approximate latitudes and longitudes could be extracted.

All latitudes and longitudes were converted to decimal format for plotting purposes and the source of each record was noted opposite the coordinates in an Excel file. The total number of records obtained per species is shown in Table 2.1.

Table 2.1: Total numbers of records for each species recorded for determining species geographic distributions in Australia.

Species common name	Scientific name	No. of records
Eastern gambusia	*Gambusia holbrooki*	4373
Redfin perch	*Perca fluviatilis*	1316
Tench	*Tinca tinca*	142
Roach	*Rutilus rutilis*	29
Yellowfin goby	*Acanthogobius flavescens*	72
Streaked goby	*Acentrogobius pflaumi*	35

Mapping: The mapping of each species involved the plotting of each record's latitude and longitude on a map of Australia. The geographic coordinate system used for displaying the maps was the Geocentric Datum of Australia, 1994 version (GDA1994). A GIS layer for the Australian stream network was provided by the Department of Environment, Water, Heritage and the Arts (courtesy of J. Wang). The location of the records was checked against this to see how closely they corresponded to known waterways. Records which clearly contained errors in the coordinates (e.g., they occurred in the sea for species that cannot tolerate high salinity, or were not close to any known waterway) were rechecked and where possible corrected. Those that could not be corrected were deleted from the record and are not included in Table 2.1. They amounted to less than 20 of the nearly 6000 total records obtained.

It was not possible to present all the detail (e.g., the occurrence of multiple records for some locations) for each species distribution in the maps produced for this report. Nevertheless, the distribution pattern of the records provides a visual estimate of each species current known geographic range in Australia and so provides an indication of the spatial extent of its distribution and hence the minimum potential scale of impacts. We also mapped the catchment boundaries for the Australian river network to show the catchments in which these alien species now occur. This layer was obtained from GeoScience Australia (www.ga.gov.au). The catchments containing each species result in a much wider geographic range than that portrayed by the dot plots, however, they indicate the likely scale for each species eventual distribution once it spreads downstream.

2.3 Species distribution maps for Australia

2.3.1 Eastern gambusia (*Gambusia holbrooki*)

All gambusia present in Australia are thought to be eastern gambusia (i.e., *Gambusia holbrooki*) and not the western species (*Gambusia affinis*) or the Dominican species (*Gambusia dominicensis*) (Lloyd and Thomasov 1985). From henceforth, gambusia refers to the eastern species, unless specified.

A recent map of this species' distribution in Australia was provided by Allen et al. (2002) and shows a general distribution across all of Victoria, ACT and New South Wales with some spread into the eastern region of South Australia and a coastal distribution in both Queensland (south of the Atherton Tableland) and around the south-western tip of Western Australia. Isolated populations were noted in the Northern Territory (in both Darwin and in the Nicholson River catchment) but no gambusia were recorded in Tasmania. Arthington and Lloyd (1989) indicate a much wider distribution, with gambusia extending northwards up the coast of Queensland to the northern tip of Cape York as well as further inland from New South Wales to include parts of the Lake Eyre drainage division. They also reported gambusia as occurring in all coastal drainages in South Australia and in rivers around the Western Australian coastline, from Geraldton north of Perth to the Pallinup River near Esperance, south east of Perth. Differences in the far northern distribution of gambusia also occurred. Arthington and Lloyd (1989) and Morgan et al. (2004) indicated that isolated populations occurred near Exmouth (on the North West cape) near, Darwin, and in the Bobby Creek catchment north of Broome. Allen et al. (2002) also recorded populations of gambusia in Darwin but not north of Broome. However, Allen et al. (2002) noted that gambusia was in the Nicholson River in the Gulf of Carpentaria, which was not recorded by Arthington and Lloyd (1989).

Our data (Fig. 2.1) indicate a wider geographic range than that portrayed by Allen et al. (2002) and Arthington and McKenzie (1997), but a narrower one than in Arthington and Lloyd (1989). Gambusia are now known to be present in the upper reaches of several eastern catchments draining into the Gulf of Carpentaria, but we found no evidence to support their occurrence in the Nicholson River. This species can be expected to spread downstream from these sites and, given the warmer conditions here and its ability to adapt to high levels of salinity, it may be able to colonise a wide range of still or slow-moving water habitats in the lower coastal regions of these catchments. We found evidence supporting the occurrence of several populations in Darwin and a discrete one north of Broome but not in the far north catchments of Queensland north of Port Douglas. If gambusia are in these catchments, we are unaware of records showing its occurrence here.

Gambusia is now also recorded from a much wider range of rivers around the Western Australian coastline. Morgan et al. (2003, 2004) and Morgan and Gill (2004) confirmed its occurrence in the Greenough, Chapman and Hutt River drainages, as well as on the North West Cape. South of Perth, it occurs in rivers draining to the west coast and extends as far east as the Pallinup River. There is anecdotal evidence (ASFB 2005) suggesting that it may now also occur in the Bremer River to the east of the Pallinup. We found no evidence to indicate its occurrence in South Australian coastal drainages apart from those around Adelaide and Port Augusta. This species is now known to occur in the Lake Eyre drainage division in South Australia and its distribution in central Australia thus conforms more to Arthington and Lloyd (1989) than to the more restricted range illustrated in Allen et al. (2002). Isolated populations of gambusia now also occur in Tasmania (Figure 2.1).

The overall distribution of gambusia in Australia indicates that it is now present throughout most of the major drainage divisions. It is present in all states and in most of the larger river systems. It currently occurs in most coastal drainages along the south-eastern coastline of continental Australia (i.e., from Port Douglas in northern Queensland, south and then east to Perth in Western Australia), but it could clearly also colonise most of the remaining coastal drainages north of Perth and east to Darwin and the Gulf of Carpentaria if there is no concerted federal and state action to halt its spread into these areas. The prospect of halting its continuing spread will present a major challenge to fishery and environmental managers over the next decade.

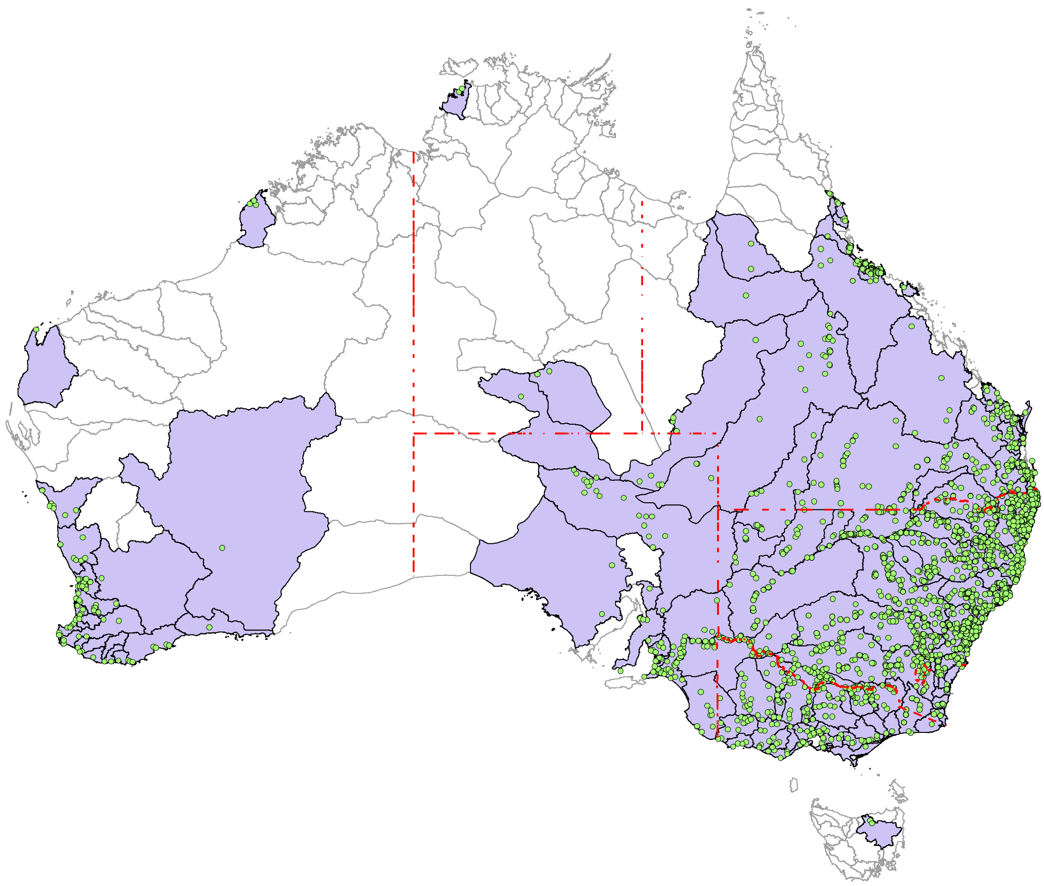


Figure 2.1: Known geographic distribution of eastern gambusia within Australia as at 2008 (green dots indicate the location of records and shaded areas are the catchments within which gambusia now occur).

2.3.2 Redfin perch (*Perca fluviatilus*)

Allen et al. (2002) indicated that redfin perch were confined to the cooler slow-flowing waters of ACT and Victoria and a large area of south-western New South Wales encompassing the Murray River system. In addition, they noted that redfin perch occurred in the lowland reaches of rivers in the south-western tip of Western Australia and in rivers in the south-east of Tasmania. The distribution in south eastern Australia was no different to that shown by McDowall (1996).

The more recent data (Fig. 2.2) also show a wide geographic distribution in south-eastern New South Wales, including major drainage divisions of the Murray River. But there are no records from most of the coastal river catchments north of Newcastle and another gap occurs in coastal catchments south of Sydney. Redfin perch have a much more restricted geographical range than gambusia, but their current distribution is likely to be determined more by anthropogenic spread than by a lack of habitable waters.

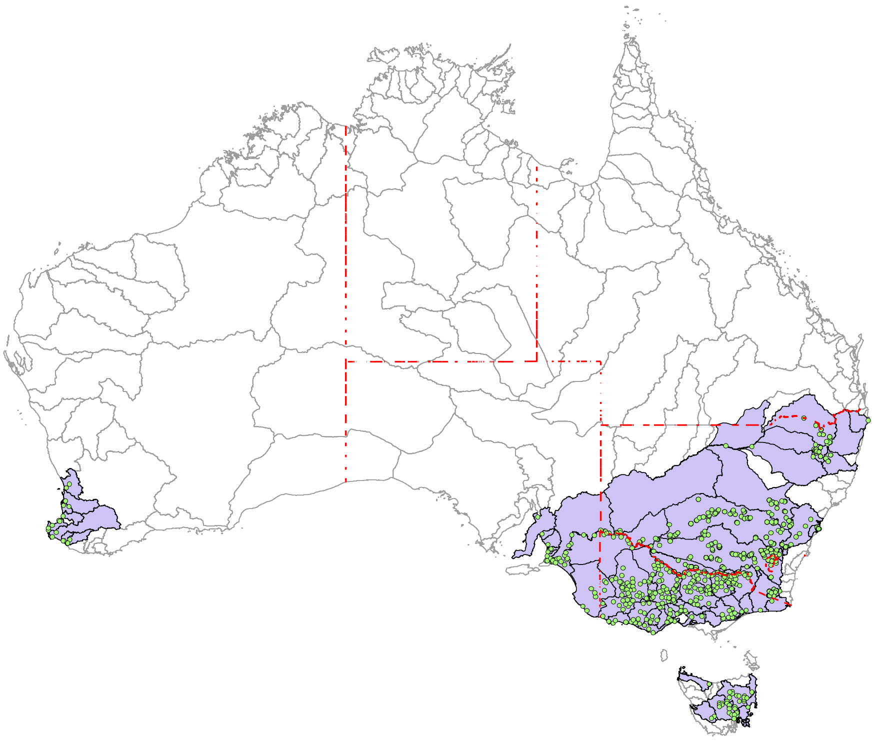


Figure 2.2: Known geographic distribution of redfin perch in Australia as at 2008 (green dots indicate the location of records and shaded areas are the catchments within which gambusia now occur).

Although Weatherley (1977) indicated that the northerly geographic distribution of perch would be limited by warm waters and that, within river catchments, their upstream penetration would be limited by high water velocities, they can clearly thrive in the relatively warm waters of Western Australia are becoming established within the northern region of New South Wales. The generally hotter waters in Northern Queensland and in the Northern Territory may well prevent redfin perch from establishing in these areas, but there is clearly considerable scope for redfin perch to be spread more widely within southern Australia and hence for the geographic scale of impacts to be increased if this species is not managed more carefully in the future.

2.3.3 Tench (*Tinca tinca*)

Allen et al. (2002) indicated that tench were present only in parts of New South Wales, in Victoria west of Melbourne and in Tasmania. Their map showed no difference in geographic distribution to that produced by Brumley (1991) and McDowall (1996). McDowall queried the presence of tench in the Murray River but data held by the New South Wales Department of Primary Industry indicates that it has been recorded in the Murrumbidgee and Lachlan Rivers which drain into the Murray. Smith and Hammer (2006) also report a localised population in the upper reaches of the Angas River which, like the Murray River drains into Lake Alexandrina near the coast. This species therefore appears to have a patchy if widespread distribution throughout the Murray River catchment. Tench have also been present in Tasmania for over a century.

The current known distribution of tench (Fig 2.3) reflects its historic distribution indicating that there has been little spread of this species in recent times. If there is any change, it may include a slight contraction in range caused by the decline of this species in waters invaded by common carp (see chapter 4).

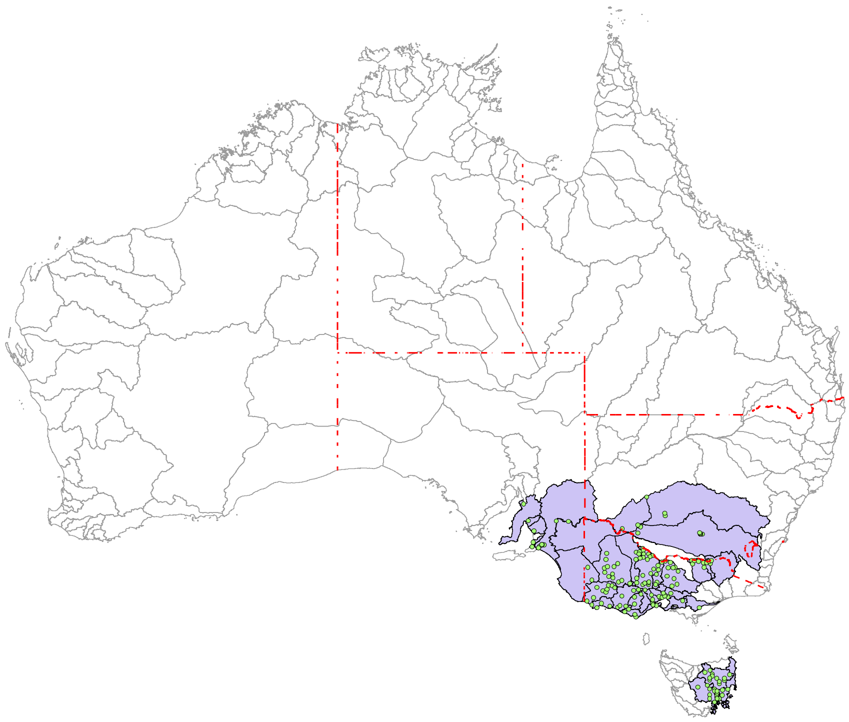


Figure 2.3: Known geographic distribution of tench in Australia as at 2008 (green dots indicate the location of records and shaded areas are the catchments within which gambusia now occur).

2.3.4 Roach (*Rutilus rutilus*)

The reported geographic distribution of roach is even more confined than that of tench. Cadwallader and Backhouse (1983) and Allen et al. (2002) both indicated that it occurred only in southern Victoria in rivers close to Melbourne.

The known geographic distribution of roach today (Fig. 2.4) is still limited but extends the distribution within Victoria presented by Cadwallader and Backhouse (1983). Roach have now been reported from Port Stephens in New South Wales (BIONET).

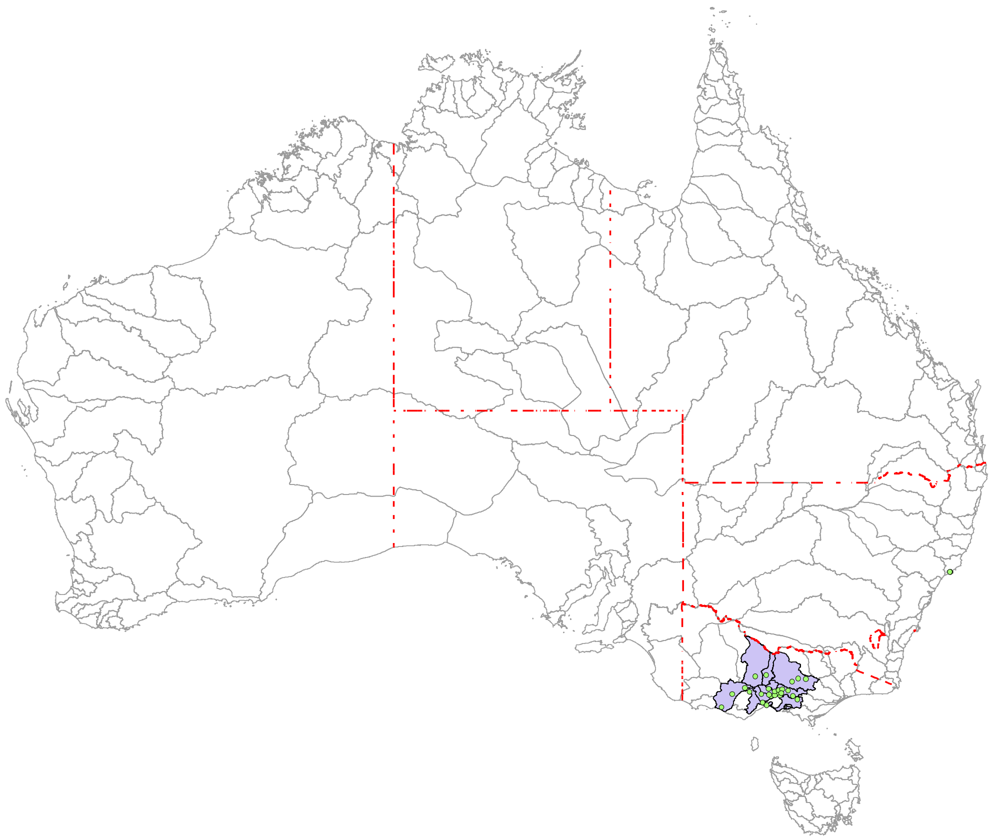


Figure 2.4: Known geographic distribution of roach in Australia as at 2008 (green dots indicate the location of records and shaded areas are the catchments within which gambusia now occur).

2.3.5 Yellowfin goby (*Acanthogobius flavescens*)

The yellowfin goby was first reported from shallow coastal waters and the lower regions of large rivers close to the Ports of Sydney and Melbourne (Cohen et al. 2001). It has since spread north and south of Sydney and occurs in the lower reaches of the Hawkesbury and Hunter Rivers to the north as well in Botany Bay to the south (Fig. 2.5). There is one record from Port Kembla, but it was not recorded here in 2000 (Pollard & Pethebridge 2002)

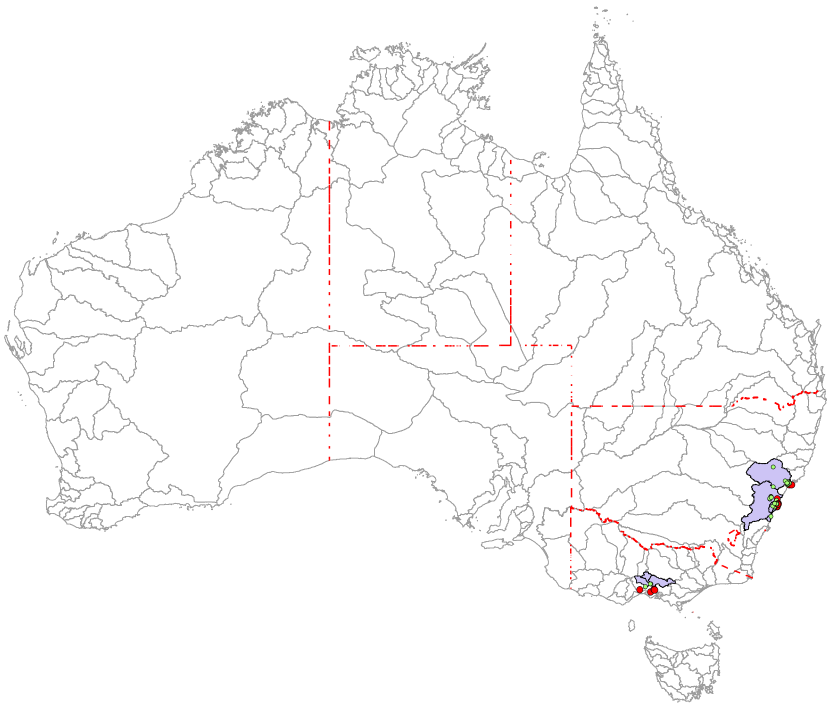


Figure 2.5: Known geographic distribution of yellowfin goby in Australian waters in 2008 (green dots indicate records from freshwater, red dots are in marine coastal waters, shaded areas indicate the catchments within which it occurs).

The yellowfin goby has not been recorded from Port Stephens which provides suitable habitat north of the Hunter River. Neither has it been reported from potentially suitable bays and inlets between Sydney and Melbourne (e.g., Port Hacking, Jervis Bay and Twofold Bay). There is also no knowledge of its occurrence in coastal waters west of Melbourne. A lack of records does not necessarily indicate absence and may merely reflect a lack of sampling. Many of these waters are also visited by ships and boats capable of spreading this species. Surveys are therefore required in waters not yet thought to be occupied by this species to confirm its absence.

2.3.6 Streaked goby (*Acentrogobius pflaumi*)

The known distribution of the streaked goby is much more restricted than that of the yellowfin goby (Fig. 2.6). It is present in Port Phillip (including Corio Bay), Sydney Harbour and Botany Bay. An isolated record from western Australia (Mead-Hunter 2005) suggested it may be present there and this has now been confirmed by the finding of a population in the Swan River (Maddern & Morrison In press). A survey of Port Kembla in May 2000 found one specimen (Pollard & Pethebridge 2002).

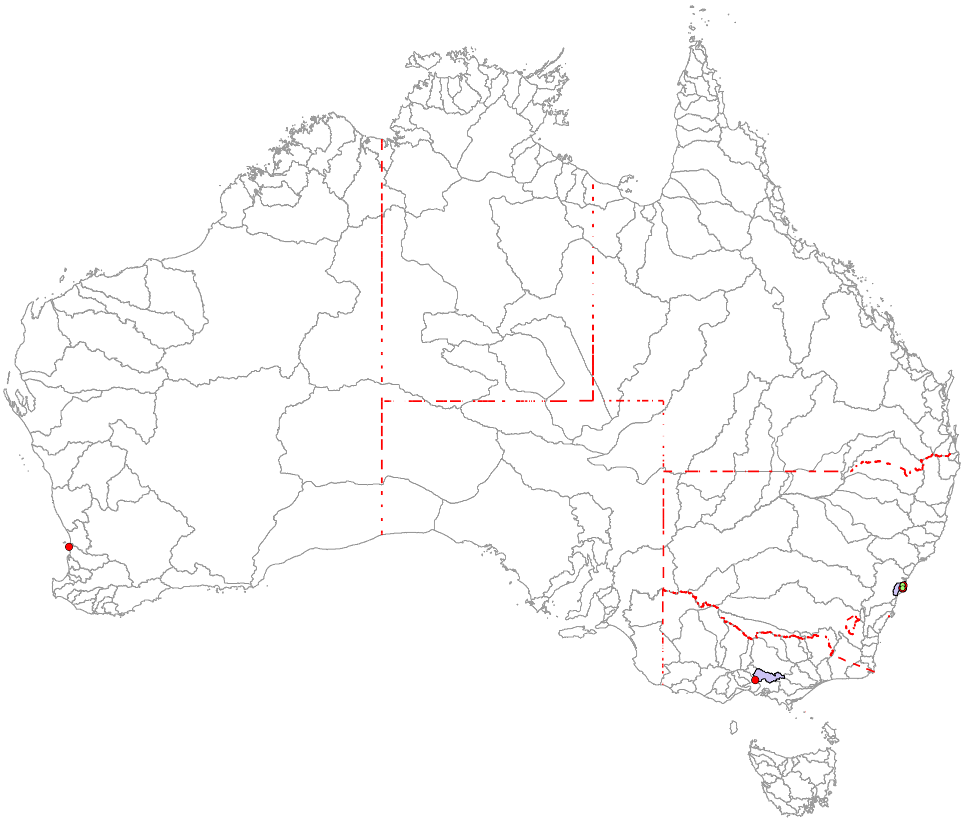


Figure 2.6: Known geographic distribution of streaked goby in Australian waters in 2008 (green dots indicate records from freshwater or estuarine waters, red dots are in marine coastal waters, shaded areas are catchments within which it occurs).

2.4 Summary and recommendations

Of the six species reviewed here, gambusia has the widest geographic distribution within Australia and it appears to be spreading, although this cannot be confirmed as the new records could be due to increased sampling coverage. This limitation in the interpretation of the data emphasizes the need for coordination of species occurrence reporting throughout Australia and for this process to indicate where certain species are absent as well as present. Confirmation of absence will depend on the sampling methods used and the habitats sampled so clearly this information needs to be recorded as well as the species found.

The currently known distribution of gambusia in Australia is at odds with its potential distribution, which, given its temperature and salinity tolerances (see Chapter 4), can be expected to encompass most of the warmer, coastal waters in northern Australia. In this respect, its potential for future spread and range expansion is high, however, the rate of spread will depend on the vectors for transport within and between catchments. Within catchments it can be expected to eventually colonise all suitable habitats downstream of sites where it currently occurs. Upstream penetration is prevented by its poor ability to cope with increased water velocities (see Chapter 4). Transfer to new catchments is likely to depend primarily on anthropogenic factors which are amenable to change through public education.

There has been no obvious increase in the broad geographical distribution of redfin perch, tench and roach over the past decade. Redfin perch has by far the widest distribution of these three species and roach the smallest. All species occur in a greater number of Victorian waters compared with their more limited distribution in other states. The geographic distribution of redfin perch in Australia is believed to be limited by a combination of warm water, high water velocities and high salinity (Weatherley 1977), however, populations are now present well inland in northern New South Wales and it is becoming more widespread in Western Australia. As with gambusia, the potential for this species to become more widespread both within and between catchments is high and will depend largely on anthropogenic factors which can be partly addressed by better public education.

The yellowfin and streaked gobies are relatively new alien fish species in Australia so their distribution is still localised and has not been mapped on a national scale before. The yellowfin goby appears to be spreading much more rapidly than the streaked goby with a number of populations now occurring up the New South Wales coast north of Sydney Harbour. However, the streaked goby has now established a population in western Australia (Swan River) and the yellowfin goby may also be present in the west. The limited distribution of these gobies may represent the 'tip of the iceberg' and it is imperative that surveys be initiated in locations where they could occur, but have not yet been recorded, to determine their current presence/absence. This will establish the geographical scale of any impacts and greatly assist with future management. Such surveys can be used to identify unaffected ‘control’ sites as against sites where these species are present (either together or alone) to serve as ‘treatment’ sites for assessment of any ecological impacts (see Chapter 3).

This report is accompanied by a CD containing the GIS files and the latitudes, longitudes and provenance of all records on the known location of the species reviewed to provide an initial, national database for DEWHA.

3. Review of impact assessment methods for alien fish

Uncertainty about the impacts of alien fish species can lead to resistance by stakeholders to contribute to the management of such fish and management by state or federal agencies will be undermined if the information on impacts is ambiguous or uncertain and the facts are in doubt (Bomford and Tilzey 1996). Unless there is good evidence of potential impacts and information on their extent, stakeholder groups may feel no obligation to contribute or assist with the process of resolving environmental issues. Consequently, impact assessment studies need to be robust and provide conclusive, unambiguous information on impacts.

The identification of the strengths and limitations of the various impact assessment methodologies used to assess the ecological impacts of alien fish is therefore a necessary precursor to any review of the evidence for impacts. Furthermore, a review of methodologies can help identify the most appropriate methods and approaches for the future. Accordingly this chapter reviews the ways in which evidence of the impacts of alien fish species has been reported in the literature, describes the main limitations that have been experienced in gathering evidence of impacts by the various methods and provides guidance on the approaches and methods that can be used in the future to answer questions on the impact of alien fish in Australia.

Opinions can vary widely about the extent and nature of impacts of alien fish species. For example, the establishment of trout in Australian waters may be seen by many as beneficial because of their recreational and sports fishery values and a lack of information on their ecological impacts. However, the establishment of common carp is regarded by most as detrimental because it serves no useful purpose and yet has a well publicised environmental impact. Such generalisations are value judgments rather than scientific assessments (Rosenweig 2001; Slobodkin 2001, cited in Lodge and Shrader-Frechette 2003) and they are based on inadequate information. As such they can present a biased view of the overall value of alien fish in Australia. A number of fish ecologists in Australia have now expressed concern over the potential for other alien fish to cause ecological problems. These ecologists provide a more coherent and rational view of the value of such fish in Australia, principally because they use a scientific approach rather than value judgements to test their concerns. However, the application of such a rational, science-based approach is currently limited by the scarcity of factual information that is available on the ecological impacts of alien fish, and this in turn can be attributed to the lack of an accepted methodology for impact assessment coupled with uncertainty over the level of proof required.

Reviews of the evidence of impacts of alien fish in Australia have been carried out by Weatherley and Lake (1967), Arthington (1991), Arthington and Blühdorn (1995), Arthington and McKenzie (1997) and Clarke et al. (2000). These studies all examined the evidence for impacts associated with a range of alien fish species now established in Australian freshwater waters and found that it was often patchy and inadequate. Koehn and MacKenzie (2004) cited a lack of information on impacts as a major barrier to future management action and Arthington and McKenzie (1997) summed up the current situation succinctly when they stated that *'there is a desperate need for hard data rather than anecdote and speculation'*. Even though the evidence of impacts was reviewed by these studies, and many of the shortcomings in information on impacts were identified, none indicated what methods are required to establish that an impact is occurring or specified the level of proof required to establish an impact. Because the level of proof required can vary greatly depending on stakeholder involvement and will help determine the methodology used for impact assessment, this issue need to be addressed first.

3.1 Establishing the 'burden of proof'

Different levels of proof can be required in impact assessments depending on the nature of the issues involved and a knowledge of the 'burden of proof' (i.e., the overall type, amount and quality of information required by managers before they can accept that an impact is occurring and therefore that an action is warranted) is important. For example, conservation groups may require a low level of proof of impact and advocate a 'precautionary approach' to alien fish control principally because there is a lack of hard information and it is better to be 'safe than sorry'. Some researchers even begin with the premise that it is rare for the introduction of alien fish to have no impacts (e.g., De Iongh and Van Zon 1993; Welcomme 1984). They assume that the mere presence of a large population of alien fish implies additional pressure on some food resource, and by implication a reduction in this for native species. Such sentiments are based on the principle that all 'niches' are filled and that the introduction of an alien species occupies a part of the niche once occupied by an indigenous species. This is the ultimate in terms of the 'precautionary approach' and it may lead to costly and unnecessary action if, in fact, there is no problem. Lodge and Shrader-Frechette (2003) suggest that we should avoid such simplistic approaches.

By contrast, groups that have some responsibility (and hence moral or financial liability) for creating or managing the impact may require a much higher burden of proof based on peer-reviewed, scientifically-defensible and replicated studies. These can be very costly, may take many years to complete and, in some cases (e.g., where control sites are lacking, or where there is a lack of pre-introduction ecological data on the environment) may be impossible to carry out successfully. Furthermore, where a fish species has a limited distribution (e.g., present at only one or two sites) replication of field trials may not be possible and so the geographical scale of impacts cannot be extrapolated. Conversely species that are widespread and contribute to recreational sport fisheries may prove to be of high social value (despite ecological impacts) in some locations but not in others. A good example of this is provided by brown trout (*Salmo trutta*) which clearly provides useful sport fisheries in some river reaches, but not in small streams that are not utilised for fishing and where the trout displace valued indigenous species (Jackson et al. 2004).

The level of proof required for acceptance of an impact can therefore vary greatly depending on the nature of the impact, stakeholder perspectives and the geographic distribution of the introduced species. Clearly some agreement is required on the level of 'proof of impact' before methodologies are selected and implemented to clarify the issues through an impact assessment. This agreement helps avoid duplication of studies and unnecessary supplementary work and it shortens the time lag before effective management decisions can be made.

Because the level of proof required to demonstrate that an environmental impact has occurred can vary greatly, it can become an issue between opposing stakeholder groups and provide a major stumbling block for managers. Some stakeholders can have entrenched views and will always find a reason to fault a methodology or change the issue to resist a change in management. Consequently, the main issue can quickly move from the ecological arena to the socio-economic one. There is no doubt that economic and social considerations are important drivers of management actions concerning fish, but they can and should be addressed within a planning context after there is agreement that an impact is occurring (or not) and not be used to delay or extend the scope and duration of impact assessment. In order to avoid such situations from developing, a mutually-agreed consensus view from all key stakeholders, or a majority agreement among them as to what the issues are and the level of proof acceptable for demonstration of an impact is desirable before the appropriate methodology for impact assessment is decided on. Once this 'level of proof' is agreed upon the methods appropriate for detecting an impact can be determined. Such an approach is novel at present in that it has rarely been applied, however, it can be expected to become increasingly important as the management of alien fish moves from a reactive to proactive phase.

3.2 Impact assessment methods for alien fish

Overall, the selection of an appropriate methodology needs to first consider the occurrence of type I and type II errors associated with data analysis and reporting. Committing type II errors (failing to detect an impact when it is present) could mean that resource managers overlook impacts and only realise that they are occurring much later, when the consequences are widespread and cannot be easily reversed. On the other hand, type I errors (identifying an impact when it is not present) could mean that money, time and resources are wasted on trying to remedy or mitigate impacts that are either non-existent or too trivial to be considered ecologically significant. The approach taken therefore needs to be selected to avoid such pitfalls, especially type II errors as these could be much more expensive to fix in the long term.

In general, the scientific literature on the impacts of alien fish such as those reviewed in this report has followed a predictable sequence of ‘impact assessment’ studies reflecting the incremental increase in knowledge gained on the nature of impacts. Initially, field studies on fish distribution and relative abundance are used to provide correlative evidence that one or more native species may have declined in locations where the introduced species is present. These initial studies are often in the nature of exploratory investigations and so may need to be repeated later by others to confirm initial observations and/or repeated elsewhere to identify the scale and generality of the suspected impact. Such studies are akin to the epidemiological approach often used to identify the effects of pollutants on human health when the mechanisms of impact are unknown.

This correlative approach typically involves a ‘before’ and ‘after’ comparison or a ‘control’ versus ‘impact’ comparison. The former assumes that some useful data are available on native fish abundance before the introduction of the alien species and the latter assumes that there will be a comparable environment where the alien species has not been introduced to that can act as a control (or reference) site. If such preliminary data and sites exist, a full BACI (Before/After, Control/Impact) approach can be contemplated. Where this can be extended in scale through the addition of more sites it can provide compelling evidence of impact even though the mechanism may not be known. However, a full BACI approach is rarely possible in practice because historic data are lacking or no suitable control sites exist. In this situation, it may only be possible to establish an inverse relationship between the presence/absence or abundance of an alien species and the relative abundance of native species. Because such relationships are correlative, and the presence of an alien species may be coincidental (e.g., it is symptomatic of change such as habitat degradation which is the main factor reducing native fish abundance), there is usually a need to establish the impact mechanism underlying the inverse relationship between an alien and native species.

In the second approach to fish impact assessment (here termed the mechanistic approach), observations from both field studies and tank studies are used to identify the potential mechanisms of impact involved (e.g., predation, competition for food or space etc.). Field studies in this phase are focused mainly on gaining basic information on the biology and ecology of the alien species in the wild (e.g., diet, reproduction, main habitats etc.) and this information is then used, in conjunction with similar data for native species, to predict whether species interactions including predation and/or competition for food or space are feasible. Such potential interactions are posed as hypotheses and are then experimentally tested in tanks, mesocosms or enclosures under more controlled conditions. In the next phase of this approach to impact assessment, field studies are carried out to demonstrate the existence of such impact mechanisms in the wild.

A third approach to impact assessment, and one which can provide more convincing scientific proof of impact, involves species manipulations to see whether the native fish recover once the alien species is reduced, and *vice versa*. In practice, deliberate species manipulations are rare because of the cost and difficulty of carrying out such field experiments on a large scale. However, recent examples in Australia include the creation of barriers to trout access in small streams and the subsequent removal of all trout above this barrier to see whether native fish recover (Jackson et al. 2004). Sometimes opportunities occur to record the results of ‘natural’ manipulations. For example, if a disease reduces the alien species (e.g., the decline of redfin perch in Lake Burley-Griffin following exposure to the EHNV virus) or if an alien fish population varies greatly from year to year because of large variations in natural recruitment.

These three approaches (correlative, mechanistic, manipulative) form the basis for ecological impact assessments of alien fish at present. Other more preliminary approaches to impact assessment such as the location of a breeding population and/or desk-top risk assessments based on species traits and invasive potential are outlined in more detail in Corfield et al. (2008).

3.3 Recommended approaches

Where the burden of proof does not need to be high, good correlative evidence of impact may be acceptable to all stakeholders even though the mechanisms may be in doubt. However, where the burden of proof is high, species manipulations may eventually be required to convince stakeholders of the cause-effect relationship between an alien species and a decline in native fauna or habitat.

Ideally, the ‘burden of proof’ will be determined for each species and agreed upon in advance by stakeholders before future studies are approved and funded. This is likely to be important for fish such as redfin perch which form an important recreational sport fishery and have a stakeholder group that may oppose any move to control it. But it may not be required for other species because they are less valued by society or, in the case of gambusia, most stakeholders have accepted that gambusia is no better than many native fish species for mosquito control.

The correlative approach is unlikely to provide a viable impact assessment method for species with a potentially high burden of proof such as redfin perch unless an inverse relationship between redfin perch and native fish can be shown for a large number of control versus treatment sites or a full BACI design can be completed. Failing this a species manipulation approach may be required but would be limited by practical considerations to small, closed water bodies. In this case, the generality of the result may be queried and manipulation experiments would be required in several widespread locations.

A less comprehensive approach is likely to be suitable to demonstrate the existence of impacts for species not requiring a high burden of proof. Although a full BACI could well be undertaken for gambusia because of the high number of sites it now occurs in, this may not be required as there is already widespread public acceptance of its impact as a result of the negative publicity focussed on it at a global scale. However, some may oppose the use of piscicides as a control or eradication tool because of concerns over the use of chemicals in natural waters and challenge the need for gambusia control at certain sites by citing a lack of unequivocal evidence on impacts. If this situation occurred, management agencies may need to provide more convincing scientific evidence of the negative impacts of gambusia and not rely on negative publicity alone. In this respect, there may be a need for several convincing studies of gambusia impact involving different species of native fish at several locations spread well apart (e.g., Western Australia, Queensland, New South Wales). This would establish the generality of impact and offset the need to establish proof of impact at each location. It would also limit the possibility of a Type I or II error occurring.

Tench and roach are generally riverine species in Australia and use of the correlative approach to detect impacts in such open environments is much more difficult than in closed environments such as lakes and ponds. A mechanistic approach is likely to prove more useful for such species, but the ‘burden of proof’ is unlikely to be as high as for redfin perch because these species lack a strong stakeholder group that could oppose control. As these fish are cyprinids and lack teeth they are not piscivorous, so predation can be ruled out. Impact assessment will therefore require a good understanding of their feeding habits, habitats, behaviour and prey species to determine the scope for displacement or reduced growth of native fish through competition for food or space. Comparable information on the distribution, feeding habitats, feeding times and prey species of native fish that may be affected by roach and tench would also be required to determine the scope for competition. Where such information can be obtained from several treatment and control sites, comparisons can be made to determine any differences in the distribution and/or growth rate of native fish that might be caused by these alien fish. It may prove difficult to establish evidence of impacts by these fish on other species, especially if impacts are minimal. If impacts are in fact minimal, the risk of a Type I error needs to be avoided.

The marine gobies have very localised distributions (e.g., Sydney and Melbourne harbours) but are apparently relatively common where they are present. Although there have been no impact studies on these species to date, they may be having a negative effect on native gobies present in these harbours (see chapter 4). Both correlative and mechanistic studies would be useful to detect these. For example, the density and distribution of native species likely to be affected could be contrasted in harbours invaded and not yet invaded by the alien gobies. At the same time, the diets and microhabitats of the alien gobies and native species occupying a similar niche could be measured to identify any potential mechanisms and explain any differences found.

A key consideration for any assessment of impacts of alien fish relying on comparisons of relative abundance is that monitoring of these parameters should incorporate appropriate capture techniques and levels of effort to ensure that the information is as robust and comparable as possible. Pilot studies may need to be carried out for some of the alien fish species to determine the most appropriate gear types for sampling them and the levels of sampling effort required to reliably detect changes in their abundance before planning impact assessments.

4. Impacts on native fauna

In this chapter, information on the biology and ecological requirements of the species in aquatic ecosystems overseas is presented as a basis for assessing their potential impact and spread in Australian waters. The sources for this review include a range of key scientific papers and reports, as well as information obtained by searching internet databases, including Aquatic Sciences and Fisheries Abstracts, the ISI Web of Science, Google scholar, Fishbase (Froese and Pauly 2007), the Invasive Species Specialist Group database (ISSG 2007), the Biota Information System of New Mexico (BISON 2003) and the United States Geological Service Exotic Fish database (Nico and Fuller 2007).

Wherever possible, information was obtained from studies in Australia. However, where this was not possible, information was obtained from overseas studies. Although information obtained from overseas sources provides a guide to the biology and ecology of a given species in Australian waters, it must be acknowledged that the Australian continent has a different climate, a different geological and topographical structure, and lacks some of the predators that would normally limit the spread, abundance and behaviour of introduced fish species in their native environments. The ecological attributes of each species considered in this report are summarised below, followed by a review of specific studies and/or reviews relevant to the introduction and potential impact of each species in Australian waters. Unless specified otherwise, the lengths of fish are the ‘Total lengths’ as against ‘Standard’ or ‘Fork lengths’.

4.1 Gambusia (*Gambusia holbrooki*)

Description, subspecies, and hybridisation: *Gambusia holbrooki* is a small (typically 10-30 mm and less than 80 mm long), fusiform fish with whitish flanks and a slightly darkened dorsal surface. The genus *Gambusia* is distinguished from other poeciliid (live-bearing) fish by the position of the dorsal and anal fins. The origin of the dorsal fin is behind the origin of the anal fin (Allen et al. 2002). Females are larger than males and when mature have a dark ‘periproctal’ spot above their vent. Males and females have a flattened head with a dorsally positioned mouth that facilitates feeding near the water surface. The conical teeth and relatively short oesophagous and intestine indicate a predatory lifestyle and carnivorous diet (Pyke 2005).

Clunie et al. (2002) and Pyke (2005) both note that the word ‘gambusia’ derives from the word ‘gambusino,’ used in Cuba to mean ‘nothing’ or ‘something worthless’ (Krumholz 1948). In Australia, gambusia has been called ‘mosquitofish’ because of its supposed effect on mosquito larvae, as well as ‘dambusia’ and ‘plague minnow’ because of its impact on amphibia and small native fish. It is now becoming known simply as ‘gambusia’ to remove the implication that it is an effective control for mosquitos.

Historically, there was some confusion as to whether the gambusia species present in Australia was *G. affinis* or *G. holbrooki*. Initially it was regarded as a subspecies (i.e., *Gambusia affinis holbrooki*) (Lloyd and Tomasov 1985) but later studies (Wooten et al. 1988; Robins et al. 1991) recognised *G. affinis* and *G. holbrooki* as distinct species respectively termed western and eastern gambusia to reflect their natural distribution in North America. Taxonomic studies of specimens from across most of Australia have now confirmed that the gambusia in Australia is *G. holbrooki* (Pyke 2005).

Hybridisation is possible (Pyke 2005). Hubbs and Lagler (1958) reported that 'intergrades' between *G affinis* and *G. holbrooki* were introduced into Michigan, and Dill and Cordone (1997) noted that a hybrid was introduced into California. However, hybrids rarely occur in natural populations (Wooten and Lydeard 1990). Lloyd and Tomasov (1985) examined specimens of gambusia in the South Australian Museum thought to be *G. dominicensis* and concluded that, although they had lower lateral line scale counts, they were *G. affinis holbrooki*, and so were subsequently confirmed as *G. holbrooki*. However, no specimens of gambusia were obtained by Lloyd and Tomasov (1985) from the vicinity of Alice Springs where *G. dominicensis* may still be present (Allen et al. 2002).

The introduction of *G. holbrooki* to southern Europe for the control of mosquito larvae in the early 1900s resulted in a genetic bottleneck and a strong reduction in genetic diversity of the European populations relative to North American populations (Grapputo et al. 2006). Despite this large reduction in its genetic diversity, *G. holbrooki* successfully invaded many European waters. Its capacity to overcome such genetic bottlenecks is reflected in its ability to readily and quickly adapt to new environments through genetic heterogeneity. For example, Langerhans et al. (2004) found adaptive differences in body morphology and swimming speeds between populations inhabiting environments with and without piscivores. Such adaptive changes may reflect genetic differences because allele frequencies have been found to differ both between and within populations (Pyke 2005). Trendall (1982; 1983) found wide variation in the life-history traits of populations of *G. holbrooki* in Western Australia, indicating an inherent genetic plasticity and ability to rapidly adapt to changing environments.

Indigenous range: *Gambusia holbrooki* occurs naturally in the east coast drainages of North America from New Jersey down to Florida and southern Alabama (Wooten and Lydeard 1990; Froese and Pauly 2007). Its latitudinal range is 40°N to 31°N compared with 42-26°N for the more westerly distributed *G. affinis* (Froese and Pauly 2007).

Habitats and migrations: *Gambusia affinis* is found mainly in the warm, shallow, marginal waters of lakes, ponds, swamps and wetlands and in the weedy margins of streams and rivers (Casterlin and Reynolds 1977). In New Zealand, it occurs in the saline, mangrove-swamp habitats present in harbours and estuaries (Rowe et al. 2007). It prefers still to flowing water habitats and Plaut (2002) found that water velocities up to 25 cm/s limit its swimming ability. Like *G. affinis*, *G. holbrooki* is a poor swimmer and prefers still waters to flowing waters. It does not tolerate fast-flowing or turbulent waters, and water velocity barriers formed by rapids, weirs, chutes and falls limit its upstream penetration. It is rarely found in fast-flowing waters (Lloyd et al. 1986), and in slow-moving reaches of rivers and streams is confined to the still-water microhabitats along the margins. In Australia, gambusia populations also occur in salt lakes, thermal springs and the cooling ponds of thermal power stations (Arthington and Lloyd 1989).

Gambusia prefers shallow waters (generally less than 15 cm) and it often occupies waters that are too shallow (1-5 cm deep) for other fish. Pyke (2005) noted a preference for waters 5-15 cm deep, which is in accordance with its habit of foraging near the water surface. Pyke (2005) also noted preferences for dark over light substrates (for the smaller fish), and for areas without any floating vegetation. In Australian waters, gambusia frequent the edges of pools and open waters in the shallows (Arthington 1988) and larger individuals are associated more with the bottom of macrophyte beds than the surface regions (Stoffels and Humphries 2003). Lloyd (1984) indicated that gambusia may burrow into the mud to avoid icing over in winter.

Gambusia is not known to undertake active migrations, although there may be some seasonal movement mainly by over-wintering females to the deeper-waters of lakes and ponds during late autumn (Pyke 2005). For example, gambusia is rarely found in shallow habitats over winter months but has been observed near the bottom of deep pools in winter. Downstream movement is probably caused mainly by floods (Haq et al. 1992) and Congdon (1994a) found that large female gambusia showed a preference for downstream movement compared with smaller fish, consistent with the role of large females as the principle colonisers of new habitats (Robbins et al. 1987). In terms of its behavioural characteristics, *G. affinis* has a greater dispersal ability than *G. holbrooki* (Rehage and Sih 2004).

Tolerances and limiting factors: *Gambusia holbrooki* is reported to occur naturally in locations where summer water temperatures range from 15-35°C, versus a lower range of 12-29°C for *G. affinis* (Froese and Pauly 2007). In his comprehensive review of these two species, Pyke (2005) noted that the temperature preference for *G. holbrooki* was slightly higher than that for *G. affinis*. Similarly, the critical thermal maximum temperature for *G. affinis holbrooki* was 38°C (Al-Johany and Yousuf 1993) compared with 35°C for *Gambusia affinis* (Otto 1973). Al-Johany and Yousuf (1993) found that the preferred temperature for *G. affinis holbrooki* increased with acclimation, whereas Winkler (1979) indicated that the preferred temperature for *G.*

affinis was 31°C under both laboratory and field conditions. Thus, a number of independent studies all suggest that *G. holbrooki* tolerates warmer waters than *G. affinis*. The maximum reported temperature tolerated by *G. affinis* (for periods of hours) was 44°C (Cherry et al. 1976).

Arthington et al. (1986) calculated a critical minimum temperature for *G. holbrooki* of 1.8°C and Pyke (2005) found studies reporting the survival of gambusia (either species) at temperatures of 1°C for brief periods and at 3-5°C for longer periods (days). Little is known about the long term survival of gambusia under cold conditions; factors other than water temperature (e.g., the size of fish and their over-wintering fat reserves) may also influence their long term winter survival. Otto (1973) found that the lower lethal temperature for *G. affinis* acclimated to cold temperatures was 0.5°C. However, it is apparent that gambusia can survive under ice in some locations, but not in all, nor at all times (Pyke 2005).

The wide temperature tolerance for *G. holbrooki* indicates that it is a eurythermal species. However, its relatively high preferred temperature indicates that it is also a warm-water species. Low water temperature is reported to limit both growth and reproduction in *G. holbrooki* (Pyke 2005). In the more southern latitudes of Tasmania, *G. holbrooki* only started growing after water temperatures exceeded 19°C and no growth occurred at temperatures less than 15°C (Keane and Neira 2004). The optimum temperature for the growth of *G. holbrooki* recorded in a number of studies was 25°C (Pyke 2005). The high water temperatures occurring in the more northern latitudes of Australia can be expected to result in fast growth and population expansion.

The onset of reproduction in gambusia in both Tasmanian and Western Australia waters also appears to be limited by water temperatures and begins once temperatures are over 15°C (Medlen 1951; Pen and Potter 1991; Keane and Neira 2004). However, the seasonal timing of reproduction is also influenced by photoperiod (Pyke 2005). Pen and Potter (1991) indicated that a minimum day length of about 12.5 hours was required.

Gambusia tolerates brackish and salt water, but not the abrupt transfer from fresh to salt water (Nordlie and Mirandi 1996; Congdon 1994b). Mortality rates increase depending on the overall difference in salinity and the extent of acclimation. Al-Daham and Bhatti (1977) found that good long term (>24 hr) survival (>90%) occurred when gambusia were transferred from freshwater to 20 ppt seawater, but not to 31 ppt seawater. However, gambusia have been reported to live in concentrations of NaCl up to 30 g/l (i.e., a salinity of c. 30 ppt.) in salt lakes (Chessman and Williams 1974) and Morgan et al. (2004) recently found populations surviving in water with a salinity of 60 ppt, which is close to the LC_{50} level (Chervinski 1983). Congdon

(1994b) found genetic differences in salt tolerance among different populations of gambusia and populations living in such high levels of salinity can be expected to have adapted to them.

Pyke (2005) reported that gambusia can tolerate oxygen levels as low as 1 mg/l, without gulping air at the water surface. They have been found to survive concentrations as low as 1.3 mg/l when held in cages beneath the water surface so that air gulping was not possible (Odum and Caldwell 1955). Below 1 mg/l, mortality increases unless air gulping at the surface is possible.

Froese and Pauly (2007) indicated that the pH tolerance for gambusia was 6-8, and that the tolerance of gambusia to a wide range of chemical pollutants and biocides is relatively high compared with other fish species (Pyke 2005). Lloyd et al. (1986) also provide a comprehensive summary of tolerance studies on *G. affinis* to a wide range of organic chemicals.

Predators, parasites and diseases: No predators are reported for *G. holbrooki* in Froese and Pauly (2007), but the morphologically similar and closely related *G. affinis* is preyed upon by eels (*Anguilla australis*), bass (*Micropterus salmoides* and *Morone saxatilis*) and cichlids (*Cichla ocellaris*) (Froese and Pauly 2007). In its natural habitat, *G. holbrooki* is preyed on by the red swamp crayfish (*Procambarus clarkii*) and Leite et al. (2005) showed that *G. holbrooki* did not display anti-predator behaviour in the presence of crayfish. In North America, gambusia is a prey species for piscivorous fish such as catfish and bass, for birds including herons, egrets, bitterns, grebes, ducks, and kingfishers, for some snakes and for predatory invertebrates such as backswimmers, water boatmen, diving beetles and dragonfly larvae (Meffe and Snelson 1989; Swanson et al. 1996). In Western Australia, *G. holbrooki* forms a major component of the diet of marron (*Cherax cainii*), and the yabbie (*Cherax destructor*) (Beatty 2006), which was introduced from south-western Australia. Morgan et al. (2002) found that redfin perch consumed gambusia and Boulton and Brock (1999) found that the little black cormorant (*Phalacrocorax sulcirostris*) fed mainly on alien fish including gambusia. Lloyd (1984) indicated that gambusia was a prey species for native fish including eels (*Anguilla* sp.), gudgeons (*Mogurnda and Gobiomorphus* sp.), the spangled perch (*Leiopotherapon unicolor*), and the mouth almighty (*Glossamia aprion*). Gambusia was also eaten by the water rat (*Hydromus chrysogaster*) and the fish eating bat (*Myotis adversus*) (Lloyd 1984). Arthington et al. (1986) found that *G. aprion* readily fed on gambusia, taking the smallest fish first. Water spiders are apparently able to capture and feed on small gambusia (Suhr and Davis 1974). In New Zealand, *G. affinis* is preyed on by eels (Chisnall 1989), and manipulation experiments have shown that stocked rainbow trout (*Oncorhynchus mykiss*) reduce gambusia populations in some lakes, either by predation or by restricting populations to inshore areas where marginal cover, such as reed beds or macrophytes, occurs (Rowe 2003). Similar observations on the effects of trout and perch on gambusia have been made by Molony et al. (2005). Interestingly, Rehage et al. (2005b) found that *G. holbrooki* reacts to the presence of large piscivorous fish by reducing activity and food consumption whereas *Gambusia affinis* does not. *G. holbrooki* may therefore be less susceptible to fish predation than *G. affinis*.

At least 23 ‘natural’ pathogens have been reported for gambusia in North America (Lloyd 1987, cited in Arthington and Lloyd 1989). Gambusia are a host to the (alien) Asian fish tapeworm, *Bothriocephalus acheilognathi* (Dove et al. 1997) and *Gambusia affinis* infected with the parasitic nematode worm *Eustrongyloides ignotus* were more susceptible to predation by piscivorous fish than non-infected fish (Coyner et al. 2001).

Age, growth and size: The sexes are morphologically different and the maximum size reported by Froese and Pauly (2007) is 35 mm for males and 80 mm for females. Comparable data provided by the ISSG (2007) database are 35 mm and 60 mm, respectively. The difference in the maximum reported size for females between these databases reflects the variable information available and failure to distinguish between the species. The maximum theoretical age estimated from growth parameters is 5.1 years (Froese and Pauly 2007), but a more realistic maximum age of 15 months is reported in the ISSG (2007) database. This lower estimate is supported by Wakelin (1986) who reported a mean age of 10 months for a lacustrine population. However, some females that are hatched in late summer will overwinter and not mature until the following summer. As gambusia only breeds during one season (Pyke 2005), overwintering females can be expected to be close to 12 months old when they reproduce and to die at about 15 months old.

After reviewing a number of studies on the growth of gambusia, Pyke (2005) indicated that, in general, both male and female gambusia grow at approximately 1-2 mm per week until they reached maturity after which growth slowed especially for males. In the cool, temperate climate of Tasmania, overall growth rates were 2.8 mm/week for females and 1.3 mm/week for males (Keane and Neira 2004). The maximum length for gambusia in Tasmania was 24 mm for males and 32 mm for females; less than the maximum size of 35 mm for males and 60 mm for females reported by (Pyke 2005). Pen and Potter (1991) found that gambusia in the cooler waters of south-western Australia were larger than in Queensland suggesting that fast growth in warmer waters may result in maturation at an earlier age and hence smaller size. In Tasmania, some male gambusia were mature at 16 mm and all by 20 mm, whereas some females were mature at 22 mm and all by 28 mm.

Feeding and diet: Gambusia rely mainly on sight to detect and capture their prey (Swanson et al. 1996) and their dentition indicates that they are carnivorous, feeding on a wide range of small aquatic animals and terrestrial species that fall onto the water surface. Their principal food source is aquatic invertebrates (including small insect larvae) such as caddis, mayfly, midges and mosquito larvae) and zooplankton (Cadwallader 1979; Bence and Murdoch 1986; Arthington 1988; 1989a; Mansfield and McArdle 1998; Garcia-Berthou 1999; Margaritora et al. 2001; Blanco et al. 2004). Being opportunistic carnivores, they also feed on the eggs and small larvae of amphibia, and they are cannibalistic (Pyke 2005). The smaller males require a smaller food ration than females and they reduce foraging and growth once they are sexually mature (Krumholz 1948). Hence, the much larger females can be expected to have a wider diet and a greater overall impact on food resources than males (Rehage et al. 2005a). Pyke (2005) reviewed a number of studies on the diet of gambusia and concluded that, although they feed on a very wide range of small animals, they select prey items based on both movement and size, with larger fish selecting larger prey. He also indicated that mosquito larvae were generally a small proportion of the prey consumed and were less preferred than annelids and crustacea.

The dietary and behavioural flexibility of gambusia is well illustrated by Blanco et al. (2004). They carried out experimental studies on the diet of *G. holbrooki* and found that, in macrophyte dominated environments, *G. holbrooki* fed mostly on zooplankton and plant-associated animals in the water column. However, in open environments that lacked macrophyte cover, feeding was more focussed on zoobenthos and detritus. An increase in gambusia abundance resulted in a dietary shift from benthic feeding to feeding on zooplankton in the water column; and a change in size-segregated prey selection, with smaller fish (males and juveniles) feeding mainly on ostracods, detritus and rotifers, and larger fish feeding more on ostracods.

Maturation, spawning and fecundity: Pyke (2005) has comprehensively reviewed the reproduction of gambusia. Males have an elongate and modified anal fin that forms a gonopodium, and use this to inseminate females. As females can store sperm over their lifetime (Krumholz 1948), several broods may be fertilised and incubated from a single insemination. Gambusia are oviparous fish: the eggs hatch within the brood pouch, with larvae being released directly into the environment.

The breeding season includes spring, summer and autumn months, and is maximal in mid-summer when water temperatures are highest (Pyke 2005). There is little evidence of reproduction occurring if water temperature remains below 16°C year round (Medlen 1951). Water temperature therefore influences the seasonal timing and duration of the breeding season. However, photoperiod is also important. For example, juveniles first occurred in November in Tasmania and Western Australian waters but much earlier (in August) in the warmer (17-18°C) waters of Queensland (Keane and Neira 2004). Although insemination of females, egg incubation and hatching can occur throughout spring, summer, and autumn months, water temperatures over 16°C and daylengths over 12-13 hours are required to initiate such events (Pyke 2005). Even in geothermally heated waters where temperatures are high all year round, there is still a distinct breeding season (Pyke 2005); and even if water temperatures are favourable, reproduction is reported to cease once daylength falls below 13 hours (Brown and Fox 1966). Thus, both water temperature and photoperiod influence reproduction.

Females are reported to carry on average 40-60 eggs in a brood (Froese and Pauly 2007) and the maximum reported is 375 (Pyke 2005). The number of eggs in a brood is dependent on a number of factors (e.g., fish size, time of year, water temperature) and varies markedly both within and between populations. For example, Howe (1995, cited by Pyke 2005) reported 50-100 eggs per female depending on fish size, with larger females producing more eggs than smaller ones. In Tasmania, Keane and Neira (2004) reported a range of 3-144, with a mean of 55.5, whereas Pen and Potter (1991) reported a range of 8-237 with a mean of 47.4 for the Collie River in Western Australia. In the warmer and more northern waters of southern Queensland, brood size ranged from 3-108 with a mean of 22.8 (Milton and Arthington 1983). In Australia, brood size appears to be higher in the lower latitudes and lower in the higher latitudes. However, a low brood size in warmer waters may be offset by an increase in the number of broods possible per year. Mature females generally die after the summer reproductive season (Krumholz 1948), and life-time fecundity for gambusia in Brisbane is estimated to be around 205 (Lloyd et al. 1986).

The gestation period for gambusia is mainly dependent on water temperature and can range from 15-50 days (Pyke 2005). In Australia, it ranged from 21-28 days in warm, northern waters (Cadwallader and Backhouse 1983) and up to 34 days in the colder waters of Tasmania (Keane and Neira 2004). There is a delay of about 2-14 days between the birth of a litter and the fertilisation of the next, even though the female may already be carrying viable sperm when a litter is hatched (Pyke 2005). Given a gestation period of 3-5 weeks, and a delay between birth and fertilisation of the next litter of about a week, gambusia could theoretically reproduce every 4-6 weeks and produce multiple broods per year (Pyke 2005). The theoretical maximum number of broods per year is nine (Pyke 2005), which matches that observed in the wild (Milton and Arthington 1983). At present there is little knowledge of the lifetime fecundity of gambusia (Pyke 2005). Females generally die in autumn following the summer in which they reached maturity (Krumholz 1948; Pen and Potter 1991). The mean number of broods per year multiplied by the mean clutch size indicates total life-time fecundity for a population. Maglio and Rosen (1969) calculated that 10 adult females could theoretically result in the production of a population of 5 million fish in six months. In reality, population size would be limited by food and mortality. Froese and Pauly (2007) indicated a population doubling time for gambusia of about 15 months.

Population size and structure: Large aggregations of gambusia (hundreds per m^2) can occur in the surface waters of some lakes and ponds over summer months (Fig. 4.1). In riverine habitats and rice fields, densities were lower, with maximum reported levels up to 3 per m^2 (Reed and Bryant 1974; Pen and Potter 1991). In all environments, sex ratios tend to favour females (Froese and Pauly 2007; Keane and Neira 2004). Large females tend to co-occur with large males (and vice versa) because the large females prefer to associate with the larger males (McPeek 1992). However, Pilastro et al. (1997) found that small males have a copulatory advantage over larger males and noted that this may explain male dwarfism in poeciliids.

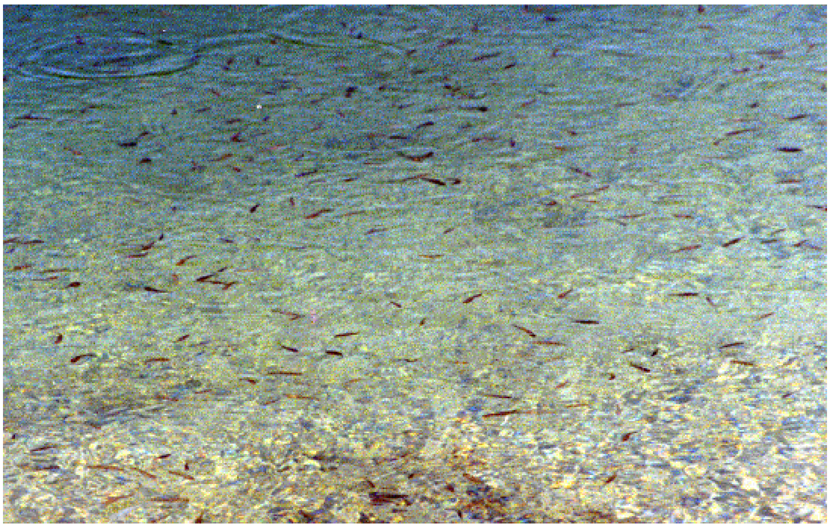


Figure 4.1: During late summer, high densities (50-100/m^2) of adult *G. affinis* occupy the entire water column in the shallow (20-50 cm deep), marginal waters of Lake Waikere, Northland, New Zealand.

Most populations of gambusia contain a mix of adult males, adult females, immature males and other fish (sex indeterminate), with the proportions varying throughout the year (Keane and Neira 2004; Pyke 2005). The general dominance of females is not caused by a size-related change in sex (as occurs in some fish species), because microscopic differences between the sexes are apparent in the internal reproductive organs from the time of birth (Pyke 2005). Because sex ratios at birth are close to 1:1 (Vargas and de Sostoa 1996), a higher mortality rate among the generally smaller males probably accounts for the female bias in adult sex ratios.

Uses: During the 1920s to the 1940s, gambusia was widely introduced throughout the world as an effective biological control agent for mosquitos. This was because of this species' ability to prey on mosquitoes at the aquatic larval stage. However, its introduction to other countries, including Australia, did not take into account the fact that native fish may already control mosquito larvae; nor that the introduction of gambusia might have an impact on some native fish species. Gambusia is now recognised as a pest species in many countries because of its impact on native fish, and native fish are often recommended for the control of mosquito larvae. This does not preclude the use of gambusia in situations where native species cannot survive, but the benefits of introducing gambusia into such locations needs to be balanced against the risk of spread and damage to endemic biodiversity. The use of gambusia for this purpose is no longer supported by any Australian jurisdiction and it will soon be listed on the national noxious fish list.

Impacts overseas: Most studies on the impacts of gambusia on native fish have been carried out in countries other than Australia, and have concerned *G. affinis*. It is therefore important to first consider the differences between these species, and how these may influence the extrapolation of impacts from *G. affinis* to *G. holbrooki* in Australia.

Froese and Pauly (2007) used population models to indicated that, in general, *G. holbrooki* are longer lived (5.1 vs. 2.3 years) and mature at a larger size and hence an older age (2 vs. 0.8 yrs) than *G. affinis*. In a study of the dispersal behaviour and invasiveness of four gambusia species, Rehage and Sih (2004) found that *G. affinis* had a greater dispersal tendency than *G. holbrooki*. *G. holbrooki* also tends to inhabit warmer waters than *G. affinis* (15-35°C vs. 12-29°C). In most other respects (e.g., size, fecundity, diet, habitats, reproductive rate) *G. holbrooki* is very similar to *G. affinis*. One study, for example, found that both species had similar feeding rates and diet breadth, and that these differed from those of non-invasive gambusia species (Rehage et al. 2005a). Hence, these two species are generally considered similar in terms of their impact on aquatic ecosystems (Courtenay and Meffe 1989). The differences in age, size at first maturation, and dispersal ability mean that it may take longer for adult populations of *G. holbrooki* to develop and spread, but once they do, their impact on native fauna is expected to be similar to that of *G. affinis*. *G. holbrooki's* tolerance of warmer water temperatures indicates its potential to colonise the most northern waters of the Australian continent.

According to Fishbase (Froese and Pauly 2007), *G. holbrooki* is known to have been introduced to 21 countries. Effects of such introductions were unknown or not documented in 18 countries. Of the three countries where some documentation was noted by FishBase (i.e., Ethiopia, India, Australia), adverse effects on the native fauna have been recorded. However, the effects of *G. holbrooki* in India and Ethiopia were based primarily on anecdotal observations. It should be noted that, although FishBase provides a global overview, it does not provide a comprehensive account of the literature on impacts and so provides a starting point to which other information must be added.

The most comprehensive studies on gambusia impacts to date have occurred in the USA and Australia. The US Geological Service fact sheet on *Gambusia holbrooki* (Nico and Fuller 2007) notes that it has been introduced to at least 33 states in North America, including Hawaii. Its northern distribution in the USA is limited by cold water temperatures, with over-wintering in some of the colder regions requiring the presence of warm groundwater springs. Winter mortality in the colder regions is high (99 %). These observations imply that winter temperatures may determine whether populations can become established. Hence, climate warming, especially in winter months, may increase the likelihood of populations becoming established further north. For example, *G. affinis* has increased its range in Missouri over the past 50 years (Pflieger 1997, cited in Nico and Fuller 2007).

In mainland USA, gambusia (species not distinguished) has caused a habitat shift in, reduced the abundance of, displaced, or eliminated local populations of at least five native fish species (Nico and Fuller 2007). Similar effects on fish have also been noted in Southern Hemisphere countries. For example, in New Zealand, gambusia was primarily responsible for the decline of dwarf inanga (*Galaxias gracilis*) in several northland dunes lakes (Rowe 1998; Rowe 2003); and for a shift in the distribution of *Galaxias maculatus* to deeper waters (Rowe et al. 2007). Adverse impacts of gambusia (mainly *affinis*) on small native fish have also been recorded in many other countries (c.f., Courtenay and Meffe 1989; Marsh and Minkley 1990; Lydeard and Belk 1993; Rinco et al. 2002; Mills et al. 2004). Competition for food was demonstrated by Caiola and de Sostoa (2005). They found no evidence of agonistic behaviour and concluded that gambusia reduced the abundance of two toothcarp species (*Valencia hispanica* and *Aphanius iberus*) through competition for food alone. However, not all small native fish are affected by gambusia; prediction of impacts depends on better knowledge of the species ecology and of the mechanisms involved.

Apart from adverse impacts on certain species of native fish, gambusia is also implicated in the decline of indigenous amphibians, especially frogs. Gambusia are known to feed on amphibian eggs and tadpoles and have reduced populations of frogs including *Rana chiricahuensis* (Nico and Fuller 2007), and *Litoria castanea*, *Litoria raniforms* and *Litoria aurea* (ISSG 2007).

Gambusia also reduces the abundance of certain key aquatic invertebrate species in freshwater environments. It has been shown to reduce the abundance of damselflies in Hawaii (Englund 1999), and dragonfly larvae, a top invertebrate predator in aquatic ecosystems, in New Zealand lakes (Rowe 1987). Several experimental studies have also indicated that gambusia can reduce populations of water beetles and back swimmers (e.g., Walters and Legner 1980 as cited by Pyke 2005, Hurlbert and Mulla 1981).

Hurlbert et al. (1972) first noted the ability of gambusia to reduce zooplankton populations in shallow waters. Later studies have confirmed this (e.g., Meiro et al. 2001) and, in some cases, have clearly linked gambusia predation on zooplankton to increased phytoplankton and reduced water clarity in pond and lake ecosystems (e.g., Nagdali and Gupta 2002). Such effects often occur when zooplanktivorous fish species reduce large cladocerans in the plankton to the extent that control over phytoplankton is relinquished and algal blooms occur, reducing water clarity. Associations between high densities of gambusia and blue-green algal blooms have been noted for ponds in Italy (Margaritora et al. 2001). It was found that large cladocera were rare in a temporary pond when gambusia was present but abundant when it was not. Similarly, Blanco et al. (2004) confirmed the preference of gambusia for large zooplankton when gambusia density increased, and when the presence of some macrophytes afforded cover from predators. In India, the mass mortality of gambusia in a shallow lake (caused by fungal infection) resulted in a doubling of zooplankton density, a 50% reduction in phytoplankton, and a marked improvement in water clarity (Nagdali and Gupta 2002). When the gambusia population recovered, the lake returned to its usual phytoplankton-dominated, turbid state.

Introduction and impacts in Australia: Wilson (1960) documented the introduction of gambusia to Australia, and Clunie et al. (2002) provide a comprehensive account of both the initial introductions and subsequent spread of gambusia in Australian waters. The species was first imported from North America to Sydney in 1925. Introductions to Brisbane also occurred in 1925 and to Cairns in 1926. In 1934, gambusia was introduced to Western Australia (Mees 1977), and after 1940, it was transported to the northwest coast and Darwin (Lloyd et al. 1986). It was subsequently spread to standing waters close to military camps throughout much of Australia (Myers 1965). By the 1980s, Merrick and Schmida (1984) reported gambusia from eight of the eleven main drainage divisions covering the Australian continent, and suggested that its range had expanded in central Australia since the 1974 floods. Lloyd et al. (1986) reported its presence in ten drainage divisions. At this time it had not been reported from Tasmania, but in 2000, gambusia was recorded in the Tamar River catchment of Tasmania (Keane and Neira 2004). It is also present in a few southern ponds in Tasmania where eradication is now being carried out (Hardie et al. 2006).

It is apparent that gambusia is now well established in Australia and that its geographic distribution is slowly expanding (see chapter 3). If its spread is not halted, it can be expected to eventually occur in all drainage divisions in Australia and, within these, to eventually colonise all suitable habitats. However, as it is a warm-water fish, its abundance is likely to be greater in the more northern, sub-tropical waters than in the colder southern zones. Here, it can be expected to occur mainly in the lowland, coastal drainages, rather than in the higher altitude, cooler waters further inland.

There is an extensive literature on the impacts of gambusia on the native fauna of Australia, particularly native fish and amphibians. However, it should be noted that while impacts on indigenous biodiversity can quickly become apparent, more subtle environmental effects are also possible and attract much less attention. For example, van den Broek et al. 2002) found that gambusia in the Manly Lagoon in Sydney contained very high zinc levels, and these can be bio-accumulated up the food chain to affect higher vertebrates including piscivorous fish and birds. Furthermore, gambusia are known to reduce dragonfly larvae (invertebrate predators) in lakes and ponds and they can also reduce water quality in shallow lakes. Such impacts can also be anticipated in Australia, but at present, there is little available evidence to confirm these here. This is in contrast to the many reports on the impacts of gambusia on fish and amphibian. These are therefore reviewed here to better understand the evidence for impacts on biodiversity and to identify the native species likely to be most affected. Impacts on small native fish are covered first and amphibia second.

A number of official state websites now provide information on the impacts of gambusia in Australia. For example, the Queensland Department of Primary Industries website notes that gambusia has been implicated in the decline of nine native fish species in Australia, belonging respectively to the genera *Ambassis*, *Chlamydogobius*, *Craterocephalus*, *Galaxias*, *Melanotaenia*, *Mogurnda*, *Pseudomugil*, *Retropinna* and *Scaturiginichthys* (Queensland DPI 2007). Similarly, the Murray Darling Basin Commission website indicates that gambusia has been implicated in the decline of 10 species of frog and nine species of native fish in Australia (MDBC 2007). The New South Wales Department of Primary Industry website states that 35 fish species have been affected by gambusia on a global basis, with gudgeon, hardyheads, pygmy perch and some rainbow fish being impacted in Australia (New South Wales DPI 2007).

Information on such websites is necessarily condensed, and often simplifies, thereby over-emphasizing, the evidence found in primary scientific publications. This process of simplification does not always recognise the complexity involved in establishing scientific proof of impact (see chapter 2) and results in the adoption of a precautionary approach. Hence it is important to examine the primary literature on which the internet-published conclusions are based.

This review of impact studies on gambusia indicates that, in general, the scientific literature follows a predictable sequence of impact assessment studies, reflecting the incremental increase in knowledge gained on the nature of impacts (see chapter 2).

Initially, field studies on distribution and relative abundance are used to provide correlative evidence that one or more native species may have declined where the introduced species is now present. These initial studies may be repeated later by others to confirm initial observations and/or to identify the scale and generality of the suspected impact. In the second phase of impact assessment, observations from both field studies and tank studies are used to identify the potential mechanisms of impact involved (e.g., predation, competition for food or space etc.). Field studies in this phase are focused mainly on gaining information on the biology and ecology of the introduced species (e.g., habitats, reproduction, diet etc.), and this information is used to predict whether species interactions including predation and/or competition for food or space are feasible. The tank studies experimentally investigate the likelihood of such interactions under more controlled conditions. In the third phase of impact assessment, field studies are carried out to demonstrate the existence of such impact mechanisms in the wild. A fourth phase, usually necessary for convincing scientific proof of impact, involves species manipulations to see whether the native species recover once the alien species is removed, and vice versa. However, species manipulations are rare because of the cost and difficulty of carrying out such field experiments. An analysis of the literature on impact assessment needs to examine the information gained for each of these four phases to determine the overall weight of evidence for impacts. This report classifies the existing major studies on the impacts of gambusia on native fish in Australia into one or more of these phases, in order to evaluate the evidence for impacts.

The first field studies noting a potential impact of gambusia on native fish were carried out in the Brisbane area by Williams (1971). He observed that a number of small, native, pelagic fish species were scarce in enclosed waters invaded by gambusia. Soon after, Sarti and Allen (1978) carried out a survey of the Northern Swan Coastal plain in Western Australia and also noted the scarcity of native fish following invasion by gambusia. Hoese et al. (1980) found that the purple-spotted gudgeon (*Mogurnda adspersa*) had declined following the invasion of gambusia, and Arthington et al. (1981) then found that Melanotaenids and Retropinnids were scarce in reaches of the Enoggera Creek where gambusia was abundant. Soon after these initial observations, Arthington et al. (1983) carried out a more systematic survey of fish abundance in the vicinity of Brisbane and found that the firetail gudgeon (*Hypseleotris galii*), the crimson rainbow fish (*Melanotaenia duboulayi*, then referred to as *M. fluviatilis*), Agassiz's glassfish (*Ambassis agassizi*, then referred to as *A. nigripinnis*), and the empire gudgeon (*Hypseleotris compressa*), were all inversely and significantly ($P < 0.05$) correlated with gambusia. However, it was not possible to attribute the decline of these species solely to the role of gambusia, due to the co-occurrence of the introduced swordtail (*Xiphophorous helleri*), as well as habitat degradation. McKay (1984) found that *M. fluviatilis* (more likely *M. duboulayi),* the

Pacific blue-eye (*Pseudomugil signifier*), some hardy heads (*Craterocephalus* sp.) and smelt (*Retropinna semoni*) were all scarce in coastal Queensland streams where gambusia was present. He noted that the abundance of *P. signifer* had declined from 73% to 1.5% of the total catch after both gambusia and *X. helleri* invaded the Brisbane River, but evidence for the decline of the other species was based on Arthington et al. (1983). Arthington et al. (1983) reported a weak and non-significant inverse correlation between gambusia and two species of *Craterocephalus*, *R. semoni* and *P. signifer*, but attributed the decline of these species more to habitat alteration than to gambusia. Nevertheless, there was some evidence from other studies (Williams 1971) that implicated gambusia in the decline of *R. semoni*.

Lloyd and Walker (1986) reported the scarcity of the southern pygmy perch (*Nannoperca australis*) and the purple spotted gudgeon (*M. adspersa*) in the lower reaches of the Murray River and attributed this to the spread of gambusia. Glover (1989) then found that gambusia was associated with a reduction of both the desert goby (*Chlamydogobius eremius*) and the spangled perch (*Leiopotherapon unicolor*) in the Clayton Bore, and three reports documented the scarcity of juvenile red-finned blue-eye (*Scaturiginichthys vermeilipinnis*) in waters where gambusia was present (Unmack and Brumley 1991; Unmack 1992; Wager 1995). In the Greenough River (WA), gambusia and the Murchison River hardyhead (*Craterocephalus cuneiceps*) were found together at only 1 of 21 sites sampled, even though they occupy similar habitats (Morgan and Gill 2004). This finding suggests interspecific competition resulting in spatial exclusion. There is now a useful account of the decline of the dwarf galaxias (*Galaxias parvus*) in gambusia infected waters on the Native Fish Association of Australia's website (NFA 2007) and, more recently, Unmack and Paras (2007) found that the Eastern little galaxias or dwarf galaxiid (*Galaxiella pusilla*), which was once widespread in Victoria, is now scarce where its distribution overlaps with that of gambusia.

These phase one field studies strongly suggest that gambusia adversely affects the abundance and distribution of a large number of the small, endemic fish species in Australia, but other factors such as changed land-use, degradation of aquatic habitat and the introduction of other alien fish species could also cause such changes (Arthington et al. 1983; 1990). It could be argued that the increased abundance of gambusia in these places is a consequence of the other changes and merely coincidental to the decline of the native species. Proof of impact therefore requires knowledge of the mechanisms by which gambusia could, or has, reduced the abundance and distribution of native fish species. In turn, this requires information on the biology, ecology and interactions of both gambusia and the affected species.

A number of key field studies on the biology and ecology of gambusia in Australia were carried out mainly by Arthington and her co-workers (e.g., Milton and Arthington 1983; Arthington 1988; Arthington et al. 1990; Arthington and Marshal 1999), but also by Pen and Potter (1991) and more recently Stoffels and Humphries (2003) and Kean and Neira (2004). This information provides a better understanding of the habitats, foods and reproduction of gambusia in Australian waters and so helps identify where and when competition with native fish could occur. It should be noted, however, that predictions of impact rely on having equally thorough knowledge of the biology and ecology of the native fish species, which is often lacking.

Many tank studies have also been undertaken to help determine the nature of interactions between gambusia and native fish. Lloyd (1990) noted that gambusia nipped the fins of other fish, and Gill et al. (1999) raised the issue of fin-nipping by gambusia on the survival of the Western pygmy perch *N. vittata*. Gill et al. (1999) used tank experiments to show that this caused a 23% mortality of the pygmy perch compared with 3% in control tanks. Howe et al. (1997) carried out long-term, outdoor tank experiments to determine the effect of gambusia on *Pseudomugil signifer*. They found that *P. signifer* stopped growing and did not mature when gambusia was present. At much the same time Koster (1997) reported results of further tank experiments involving the southern pygmy perch (*Nannoperca australis*) and gambusia. These indicated that gambusia didn't affect the growth of *N. australis* provided that food was not limiting, but that gambusia did nip the fins of both *E. australis* and *G. pusilla*, indicating scope for agonistic interactions in the wild. A series of tank studies were also carried out to identify the effects of different densities of fish on agonistic encounters. Knight (1999) examined the interaction between gambusia and *P. signifer* and concluded that the aggressive behaviour of gambusia towards *P. signifer* was positively related to gambusia density. Breen (2000) obtained a similar result when gambusia was placed in tanks with *M. duboulayi* and the ornate rainbow fish (*Rhadinocentrus ornatus*). Conte (2001) obtained the same results for gambusia and *H. galii*. However, Miles (2001) found an inverse relationship between gambusia abundance and its aggressive encounters in tanks with *A. agassizi*, as did Cronin (2001) when *H. galii* and the Oxleyan pygmy perch (*Nannoperca oxleyana*) were exposed to gambusia. Warburton and Madden (2003) investigated the behavioural interactions between gambusia and both *P. signifer* and *M. duboulayi*. They found that, in tanks, gambusia fin-nipped *M. duboulayi* but chased *P. signifer*, and that the frequency of aggressive encounters increased as food supply and fish density increased, resulting in less feeding by the native species. In a later study, Becker et al. (2005) examined the interaction between gambusia and the common jollytail (*Galaxias maculatus*). It was found that, although the spatial distribution of *G. maculatus* in tanks was changed and the mean distance to food sources increased, gambusia did not out-compete *G. maculatus* for food.

Such experiments are informative but they can be greatly influenced by the experimental conditions. For example, temperature, fish density, tank size, cover, and the type, amount and method of food supply may all play a role in exacerbating or minimising aggressive interactions (e.g., Moore et al. 2002; Becker 2005; Rowe et. al 2007). Tank experiments provide testable hypotheses about potential interactions that could occur in the wild, and so help focus field experiments; but because of the large number of variables that can influence species interactions in the wild, which cannot be easily reproduced and controlled for in tank environments, they are limited in their predictive power.

Phase three impact assessments involve the provision of field evidence of impact mechanisms. Hambleton et al. (1996) found the scales of *N. vittata* in the stomachs of *G. holbrooki* and Ivanstoff and Aarn (1999) demonstrated that gambusia eat the larvae of some native fish in the wild by finding melanotaenid larvae and one *H. galii* in the stomachs of gambusia. Native species can clearly also be reduced by mortality arising from fin-nipping by gambusia. Following on from tank experiments, Gill et al. (1999) looked for field evidence of fin damage caused by gambusia in wild populations of native fish. They found evidence of this in lentic environments, with both *N.vittata* and nightfish (*Bostockia porosa*) being affected. They also found that fin damage was greatest for younger fish, and that its incidence was reduced by cover. Faragher and Lintermans (1997) noted that fin-nipping could lead to secondary bacterial and fungal infections which result in death. Although fin-nipping can clearly reduce the densities of vulnerable native fish species, Gill et al. (1999) concluded that whereas gambusia would not cause the extinction of *N. vittata*, it had reduced its abundance and changed its distribution. The survival of *N. vittata* in this study may well have been because it could change distribution, seek cover, and so avoid gambusia aggression. In more simplified environments lacking cover or water depth, the likelihood of a localised extinction can be expected to be much higher. At present there is no confirmation of the localised extinction of a native fish species caused by gambusia. However, gambusia may have resulted in the disappearance of *Rhadinocentrus ornatus* from Eighteen Mile Swamp on North Stradbroke Island (Arthington and Lloyd 1989; Arthington 1994).

If predation (on larvae) and fin-nipping (of young fish) by gambusia doesn't reduce the densities of native fish species in the wild, interspecific aggression may still result in a changed distribution such that spatial segregation occurs. Competition for food or space can also occur and may exacerbate the effects of interspecific aggression on the spatial segregation between gambusia and native fish. For example, Fletcher (1986) concluded that gambusia caused a niche shift in wild populations of the Southern pigmy perch (*Nannoperca australis*), while Fairfax et al. (2007) documented spatial segregation between gambusia and the red-finned blue eye (*Scaturiginichthys* *vermeilipinnis*). Stoffels and Humphries (2003) found that Midgley's gudgeons (*Hypseleotris sp*.) pass through a similar spatial and trophic niche to that occupied by gambusia before they become benthic, and concluded that *Hypseleotris* may be vulnerable to gambusia at this ontogenetic stage. Such field studies provide strong evidence that interactions between gambusia and native fish will occur, but they still fall short of irrefutable proof.

The fourth phase of impact assessment involves studying the outcome of pest species manipulations in the wild. Ideally these involve controlled reductions (or increases) in the abundance of the introduced species to see how the native species respond. Although gambusia have now been eliminated in a number of small ponds in Australia, these waters were either too small for native fish, or there were no before-and-after studies of native fish carried out to determine the effects of gambusia reduction on their populations.

Despite the paucity of fourth phase impact assessment studies on gambusia, there is, nevertheless, a large and growing body of evidence to suggest that gambusia do reduce the abundance and distribution of a number of small, endemic fish species in the wild through a combination of predation and interspecific aggression mediated by competition for food and the nature of the environment. The strongest evidence is provided where several independent studies report inverse relationships in the field, where a decline has followed the invasion of gambusia, and where tank and field studies both provide evidence of the impact mechanisms. Of the 23 species for which we could find some data indicating that an impact had occurred, or was highly likely to occur (Table 4.1), both field and mechanism-based evidence was available for 8 species. Evidence for adverse impacts on another 11 species was based solely on field data from distributional studies. The potential for impacts on a further 4 species was gleaned from tank experiments not supported by any field data. None of the studies reported has involved an experimental field manipulation of gambusia abundance providing unequivocal proof of impact.

Although gambusia has been noted as having an impact on the hardyhead species, *C. stercusmuscarum* and *C. marjoriae*, Arthington et al. (1983) found a weak inverse relationship between gambusia and these species and attributed the decline of *C. stercusmuscarum* more to habitat alteration than to the presence of gambusia. Similarly, Wager and Jackson (1993) indicated in their species recovery plans that gambusia played a role in the decline of the honey blue-eye (*Pseudomugil mellis*), the Yarra pygmy perch (*Nannoperca obscura*), and Ewen's pygmy perch (*Nannoperca variegata*), but we could find no primary evidence to confirm these concerns. It is therefore likely that they are based on the personal observations and knowledge of the biologists familiar with these species, and on an extrapolation of knowledge of gambusia interactions with similar native fish species, rather than reports of impacts published in the scientific literature.

Table 4.1 Summary of evidence for impacts of gambusia on Australian native fish. Studies are categorised into those providing 'field-based' or distributional evidence for an adverse effect (e.g., mutually exclusive distributions, fin-nipping, dietary overlap), and those providing information on potential mechanisms of impact by way of experimental 'tank-based' or enclosure studies. Taxonomy and nomenclature follows Allen et al. (2002).

No.	Common name	Scientific name	Studies on impacts (numbers refer to studies listed below the table)	
			Field-based	Tank-based
1	Agassiz's glassfish, olive perchlet	*Ambassis agassizi*	5	18
2	Crimson-spotted rainbow fish	*Melanotaenia duboulayi*	4, 5	17, 22
3	Purple-spotted gudgeon	*Mogurnda adspersa*	3, 8	
4	Pacific blue-eye	*Pseudomugil signifer*	7	13, 15, 22
5	Australian smelt	*Retropinna semoni*	1, 4	
6	Red-finned blue-eye	*Scaturiginichthys vermeilipinnis*	10, 11, 24	
7	Firetail gudgeon	*Hypseleotris galii*	5,16	19, 20
8	Empire gudgeon	*Hypseleotris compressa*	5	
9	Midgley's carp gudgeon	*Hypseleotris sp. 1*	21	
10	Western pygmy perch	*Nannoperca vittata*	14	14
11	Nightfish	*Bostockia porosa*	14	
12	Ornate rainbowfish	*Rhadinocentrus ornatus*	6, 16	17
13	Oxleyan pygmy perch	*Nannoperca oxleyana*		19
14	Southern pygmy perch	*Nannoperca australis*	8	12
15	Common jollytail	*Galaxias maculatus*		23
16	Dwarf galaxias	*Galaxias parvus*	26	
17	Western minnow	*Galaxias occidentalis*	14	
18	Eastern little galaxias	*Galaxiella pusilla*	25	12
19	Black-stripe minnow	*Galaxiella nigrostriata*		2
20	Edgbaston goby	*Chlamydogobius squamigenus*	11	
21	Desert goby	*Chlamydogobius eremius*	9	
22	Spangled perch	*Leiopotherapon unicolor*	9	
23	Murchison River hardyhead	*Craterocephalus cuneiceps*	27	

[1]Williams (1971), [2]Griffiths (1972) cited by Morgan el at. (2004), [3]Hoese et al. (1980), [4]Arthington et al. (1981), [5]Arthington et al. (1983), [6]Arthington and Lloyd (1989), [7]McKay (1984), [8]Lloyd and Walker (1986), [9]Glover (1989), [10]Unmack (1992), [11]Wager (1995), [12]Koster (1997), [13]Howe et al. (1997), [14]Gill et al. (1999), [15]Knight (1999), [16]Arthington and Marshal (1999), [17]Breen (2000), [18]Miles (2001), [19]Cronin (2001), [20]Conte (2001), [21]Stoffels and Humphries (2003), [22]Warburton and Madden (2003), [23]Becker et al. (2005), [24]Fairfax et al. (2007), [25]Unmack and Parras (2007), [26]NFA (2007), [27] Morgan and Gill (2004).

The evidence for impacts by gambusia on many endemic fish species may well fall short of unequivocal proof at the species level, but collectively it indicates that gambusia does pose a problem for some endemic Australian fish in some environments and, until such environments can be identified, or the spread of this species controlled, it clearly justifies a precautionary approach. A number of impact mechanisms have now been discovered for gambusia and it is apparent that the biology of a native species and the physical characteristics of its environment will modify the nature and severity of impacts by gambusia. Knowledge of such 'modifying' factors may well contribute to the design of management programmes to reduce the impact of gambusia on native fish.

Environmental conditions under which impacts on native fish will not occur, or are negligible, can be gleaned from some of the studies that have not found an impact of gambusia on native fish. For example, Morton et al. (1988) found that gambusia co-existed with both *P. signifer* and gobiids in a salt-marsh environment, and Pusey et al. (1989) noted that gambusia co-existed with all native species in the Moore, Canning and Dandalup Rivers of Western Australia. Similarly, Pen and Potter (1991) found that the high densities of gambusia in the Collie River (3.2 individuals/m^2) had little noticeable effect on *Tandanus bostocki* (freshwater cobbler), *N. vittata*, *Bostockia porosa* or *Galaxias occidentalis* because their larvae were not affected by predation. The breeding season for the native species occurred well before that of gambusia, allowing fry to develop before predation became a major factor. This conclusion was later reinforced by other studies which confirmed that there was little dietary overlap between these species (Pen et al. 1993). Another key factor identified by Gill et al. (1999) was the high frequency of large discharges in winter which greatly reduced the population size of gambusia in this river. Pusey et al. (1989) and, more recently, Chapman & Warburton (2006) indicated that high discharges during winter months flushed most gambusia out of the rivers, greatly reducing the gambusia population size. Conversely flow regulation can favour gambusia (Bunn & Arthington 2002). Such findings suggest that the interactions between gambusia and small native fish will be less severe in riverine as against still-water environments but will still depend on a range of temporal, spatial and environmental factors that collectively influence the scope for predation, aggression, and competition to occur, as well as the opportunity for native species to avoid such effects through a change in distribution.

The main mechanisms of impact on small native fish will depend on the specific species involved, but are likely to involve one or more of: predation on eggs and larvae, aggression and fin-nipping leading to spatial exclusion, and competition for food (Arthington and Lloyd 1989; Gill et al. 1999). Competition for food was demonstrated by Caiola and de Sostoa (2005). However, Ivanstoff and Aarn (1999) reported that gambusia consumed the fry of native fish species, confirming predation as a mechanism for the impact of gambusia on other fish species. Belk and Lydeard (1994) found that predation on young fish, and not competition for food, was the main reason why gambusia reduced populations of least killifish (*Heterandria formosa*). Similarly, gambusia have been found to reduce the western pygmy perch (*Nannoperca vittata*) in Western Australia by preying on their young (Hambleton et al. 1996). Both Rowe (1998), and Gill et al. (1999), found that native fish were attacked in the wild (fin-nipped) by gambusia. In one study, this aggressive or agonistic behaviour resulted in large mortalities of *G. gracilis*, but these fish were not consumed by gambusia:

aggression, rather than predation, was the mechanism of decline (Rowe 2003). Even though agonistic interactions may rarely be observed, aggressive behaviour can also result in a changed distribution of native species (i.e., interactive segregation, *sensu* Nilsson 1967). For example, both juvenile and adult *G. maculatus* were attacked by gambusia in aquaria indicating that aggressive behaviour can be expected in the wild. An analysis of *G. maculatus* depth distribution indicated that adults occurred more frequently in shallow waters (<0.5m) in locations where gambusia was absent, but that they occurred in deeper waters where gambusia was present (Rowe et al. 2007). Such results indicate a shift in distribution to deeper waters at locations where gambusia occur. More recently, Keller & Brown (In press) found niche shifts and behavioural differences between populations of ornate rainbow fish (*Rhadinocentrus ornatus*) that were sympatric and allopatric with gambusia. Although native fish distribution may clearly be changed through interactions with gambusia, the long term implications of such changes on the population dynamics of native fish are still largely unknown (Arthington and Lloyd 1989). It could be argued that a shift in distribution from a preferred to less preferred location will reduce population growth and production but this is yet to be demonstrated.

Combinations of these mechanisms may occur and the intensity of impacts can also be expected to vary between seasons and locations. Laha and Mattingly (2007) found that gambusia affected the Barrens topminnow (*Fundulus julisia*) through both predation (on fry and juveniles) and aggression (fin-nipping of adults). In a New Zealand dune lake, agonistic behaviour and fin-nipping by gambusia resulted in the mass mortality of adult dune lakes inanga (*G. gracilis*), but this occurred only in autumn, when limnetic foods were reduced and the galaxiids were forced to forage in the littoral zone (Rowe 2003), where gambusia were abundant. Pen and Potter (1991) concluded that *G. holbrooki* did not affect native fish in the Collie River because there was no evidence of dietary overlap, and the indigenous species bred much earlier than gambusia. The physical environment plays an important role in determining the nature and extent of species' interactions. For example, Morgan et al. (1996) observed that native fish were only abundant in waters containing gambusia if large amounts of instream cover were present. Altered hydrological regimes can also have a marked influence on gambusia and hence its impact on native species (Bunn & Arthington 2002). Floods greatly reduced gambusia populations in some rivers and so restricted the summer build-up of gambusia (Meffe 1984; Pusey et al. 1989; Chapman & Warburton 2006), but droughts coupled with water abstraction may exacerbate interactions by confining gambusia and native species to diminishing pools of water.

It is apparent that the impact of gambusia on native fish will depend on both the ecology and life history of the native fish species present, as well as the physical nature of the environment it is in. Hence the impact mechanism or combination of impact mechanisms can be expected to vary depending on both these factors. This variation in mechanisms (e.g., predation, food competition, agonistic-based displacement) can readily account for the differences in results between different studies and more knowledge on how such mechanisms are influenced by physical factors will help resolve the ‘apparent’ ambiguities between studies.

Although there is a weight of evidence indicating an adverse impact of gambusia on a number of endemic, freshwater fish in Australia, there is also increasing evidence that gambusia has reduced the abundance and distribution of a number of native anuran species. Gillespie and Hero (1999) produced a comprehensive review of the effects of gambusia (and other fish) on Australian frogs. Although the main mechanism of impact is thought to be predation by gambusia on eggs and tadpoles, increased mortality from tail-nipping of tadpoles is also included as a possibility. Gillespie and Hero (1999) found that the impact assessment studies followed a similar pattern to those outlined above for fish. A number of studies documented the negative associations between gambusia and frog species in the wild, whereas others used tank or cage studies and an experimental approach to identify the mechanisms of impact. A third group of studies examined potential mechanisms and the factors affecting these in the wild. As with the fish interaction studies, there were no reports of manipulation studies showing that an induced decline in the abundance of gambusia resulted in the recovery of frog abundance, or *vice versa*.

The studies reviewed by Gillespie and Hero (1999) revealed that the eggs of five species of frog tested were unpalatable to gambusia. However, Pyke and White (2000) found that gambusia attacked and ate the eggs of *L. aurea* and Komack and Crossland (2000) found that while the eggs of *L. ornatus* were consumed, those of the cane toad *Bufo marinus* were not. Gillespie and Hero (1999) reported tank and cage studies showing that the larvae of all ten species of frog that had been tested were readily preyed upon by gambusia. Fin-nipping of the larger tadpoles may also result in death through immobilisation, ensuing disease, or reduced feeding ability. Gillespie and Hero (1999) noted three studies recording the incidence of fin-nipping, and Webb and Joss (1997) indicated that although fin nipping occurred in wild populations of tadpoles, tadpoles may be able to cope with this.

Gillespie and Hero (1999) indicated that there were a large number of other animals and environmental factors, other than gambusia, which influenced tadpole mortality in the wild and which could provide plausible, alternative explanations for tadpole decline. Furthermore, a number of frog species appear to co-exist with gambusia in the wild, so some anuran species are likely to be less vulnerable to gambusia than others. Overall, Gillespie and Hero (1999) concluded that although the evidence for an impact was still unclear at the species level, especially for *L. aurea*, gambusia clearly contributed to the decline of several anuran populations.

A number of new studies have been reported since Gillespie and Hero's (1999) review. In particular, Komak and Crossland (2000) and Pyke and White (2000) found that gambusia attacked the eggs of some anuran species, but not all. Hamer et al. (2002a) investigated the role of water level and gambusia on the production of *L. aurea*, after noting that several studies (e.g., Reynolds 1995; Pyke and White 1996) had found that frogs were more at risk from gambusia in permanent as against ephemeral waters, and that *L. aurea* appeared to be more common in ephemeral than in permanent water-bodies. Moreover, Hamer et al. (2002b) found that the amount of riparian vegetation and its proximity to other occupied water bodies were stronger predictors of frog presence than gambusia. Hamer at al. (2002a) found that the tadpoles of *L. aurea* showed no anti-predator response to gambusia (unlike *Limnodynastes tasmaniensis*), and that although the tadpoles were vulnerable to predation by gambusia, declining water levels reduced overall 'mass at metamorphosis' by 30%. Gambusia did not affect tadpole feeding, and Hamer at al (2002a) indicated that gambusia predation may have restricted *L. aurea* to less productive, ephemeral environments. They suspected that, despite the co-existence of these species at a number of locations, gambusia could well have reduced the overall reproductive output of *L. aurea* by restricting major populations to ephemeral environments. Reynolds (2003) later noted that gambusia may have adversely affected several frog species in Perth's metropolitan lakes (species and number not stated), and in 2006, the Global Amphibian Assessment (GAA) webpage reported that *Litoria cooloensis* had 'virtually disappeared from Brown Lake on North Stradbroke Island following the introduction of gambusia there in 2003'(GAA 2007).

In summary, gambusia has now been implicated in the decline of at least 15 species of Australian frog. These conclusions were drawn from either tank-based experiments on tadpole susceptibility to predation by gambusia, or field-based studies examining changes in frog geographic distribution in relation to the spread of gambusia (Table 4.2). Compared with native fish, there are far fewer studies on anurans, especially field-based studies. Field studies of frog decline were backed up by tank based studies for only two of the fifteen species, otherwise the evidence for impact relied on distributional studies alone (3 species), or tank-based studies alone (10 species).

Table 4.2 Summary of studies providing information on the impacts of gambusia on native Australian anuran species.

	Common name	Scientific name	Studies on impacts (numbers refer to studies listed below the table)	
			Field-based	Tank-based
1	Common froglet	Crinia signifera		2, 5
2	Sign-bearing froglet	Crinia insignifera	7	4, 7
3	Glauert's froglet	Crinia glauerti		4, 7
4	Tschudi's froglet	Crinia georgiana		7
5	Green and golden bell frog	Litoria aurea	9, 13	8,12
6	Lesueur's frog	Litoria lesueuri		6
7	Bleating tree frog	Litoria dentata		6, 8
8	Slender tree frog	Litoria adelaidensis		7
9	Yellow-spotted tree frog	Litoria flavipunctata	1	
10	Southern brown tree frog	Litoria ewingii	3	
11	Cooloola sedge or tree frog	Litoria cooloonsis	14	
12	Spotted marsh (grass) frog	Limnodynastes tasmaniensis		6
13	Striped marsh frog	Limnodynastes peronii		5
14	Ornate burrowing frog	Limnodynastes ornatus		11
15	Moaning frog	Heleioporus eyrie		4, 7

[1]White and Ehmann (1977), [2]Williamson (1988) cited in Morgan and Buttemar (1996), [3]McGilp (1994), [4]Blyth (1994), [5]Webb and Joss (1997), [6]Harris (1995), [7]Reynolds (1995) [8]Morgan and Buttemer (1996), [9]White and Pyke (1996), [11]Komak and Crossland (2000), [12]Pyke and White (2000), [13]Hamer et al. (2002a), [14]GAA(2007).

4.2 Redfin perch (*Perca fluviatilus*)

Description, subspecies and hybridisation: Redfin perch are a medium-sized fish with a laterally compressed body form and adults have a pronounced dorsal hump behind the head. The redfin perch is characterised by 5-9 vertical, black, bands on each flank and by reddish-orange pelvic and anal fins. The rays of the first dorsal fin are made of stout spines and these, together with the opercula spines, mean that this fish can be difficult to handle. A key for the identification of Percidae in Australian waters is provided by McDowall (1996). This species is more commonly referred to as perch, or the European perch, and is similar in many respects to the yellow perch (*Perca flavescens*), native to North America. Thorpe (1977) and more recently Craig (2000) have provided a comprehensive synopsis of biological data on both species in the northern hemisphere, hence the following sections focus more on what is known of redfin perch biology and ecology in the southern hemisphere.

Indigenous range: The natural range for redfin perch is northern Europe and eastward to Siberia (Froese and Pauly 2007). It does not occur in the warmer, southern waters of Spain, Italy and Greece. The latitudinal range is 74°N to 38°N and the longitudinal range 91°W to 168°E (Froese and Pauly 2007).

Habitats and migration: Adult redfin perch prefer still and slow-flowing waters. Hence, they occur in many lakes throughout their natural range as well as the slower flowing riverine habitats such as pools, runs, eddies, backwaters and the lower regions of canals and rivers (Cadwallader and Backhouse 1983; Merrick and Schmida 1984; McDowall 1996). They also occur in brackish waters of the Baltic Sea (Froese and Pauly 2007). In lakes and ponds, adult redfin perch are more common close to large beds of macrophytes and/or emergent plants such as rushes. In rivers, they tend to occur close to objects providing cover such as logs, tree roots and fringing rush beds.

Juvenile redfin perch are more variable in their habitat and distribution, especially in static water environments such as lakes, dams and reservoirs. Some remain close to the lake or river edge and inhabit the shallow, weedy littoral zone; others become schooling and pelagic; and another group inhabit the bathypelagic zone (pers. comm., M. Czech, Institute of Hydrobiology Czech Republic). Such flexibility in habitats no doubt assists the survival of juvenile redfin perch.

Tolerances and limiting factors: Redfin perch are a cool water fish and Froese and Pauly (2007) reported a distributional temperature range of 10-22ºC, a pH range of 7-7.5 and a depth range of 1-30 m. Backhouse and Cadwallader (1983) indicated a temperature range of 8-27ºC, with water temperatures between 23-36°C being tolerated for short periods. However, Weatherley and Lake (1967) noted that survival could occur for brief periods at 30-31°C, and that this temperature level was a good predictor of their northern limit in Australia.

The redfin perch is capable of existing in brackish water environments (Froese and Pauly 2007) but they do not tolerate salinities greater than 10 ppt (Privolnev 1970).

Lake (1971) and Weatherly (1977) indicated that the geographic distribution of redfin perch in Australia was restricted to the south because of high water temperatures in the north and that, within rivers, distribution was restricted to the middle reaches because of high salinities in the lower reaches and high water velocities in the upper reaches. Lake (1967a) observed a high mortality of perch eggs when water temperatures increased rapidly from the minimum temperature at which spawning begins (about 11-12ºC) and indicated that this would also explain the northern limit on perch distribution in Australia.

Predators, parasites and diseases: In Australia, redfin perch are a host for and are affected by the epizootic haematopoietic necrosis virus (EHNV), which is also highly pathogenic to silver perch, mountain galaxias, Macquarie perch, and Murray cod

(Langdon 1990; Langdon and Humphrey 1987). Mass mortalities of redfin perch from EHNV have occurred in the Australian Capital Territory; some authorities have attributed the decline of Macquarie perch in the ACT to the virus (Lintermans 1991).

Age, growth and size: The maximum size reported to date is 51 cm and the maximum weight 4.75 kg (Froese and Pauly 2007). Lake (1971) reported a maximum length of 50 cm and a maximum weight of 10 kg for redfin perch in Australia. The maximum reported age is 22 years and the maximum age estimated from growth statistics is 24 years (Froese and Pauly 2007). Le Cren (1958) found that temperature was the main determinant of growth rate, with the number of degree days over 14°C explaining two-thirds of the variation in growth rate among redfin perch populations in the UK. Females are generally faster growing and larger than males and dominate the population. In Big Brook Dam (WA) the sex ratio of redfin perch was 1.7 females per male (Morgan et al. 2002). Because large, natural piscivores such as pike (*Esox lucius*) do not occur in New Zealand, perch populations in lakes are often characterised by large numbers of small fish. A similar situation often occurs in Australian lacustrine waters (New South Wales DPI 2007).

Feeding and diet: Shoals of juvenile redfin perch feed in shallow, open waters and in the littoral zone of lakes – mainly on zooplankton. In rivers, they tend to feed on small aquatic invertebrates in still backwaters, pools and in the slower flowing margins. As redfin perch grow, they feed more on larger prey, including benthic and mid-water invertebrates as well as small fish. Although adult redfin perch are carnivorous, they become increasingly piscivorous as they increase in size. Their protrusible mouth allows them to ingest other fish up to a third of their length. Schooling behaviour decreases with age/size such that the largest redfin perch (>20 cm TL) are usually solitary and feed more on the larger, less mobile, benthic species including fish and crustaceans. For example, in western Australian waters, redfin perch over 20 cm long switched to heavy predation on the marron (*Cherax tenuimanus*) (Pen and Potter 1992; Morgan et al. 2002).

Maturation, spawning and fecundity: Redfin perch age at maturation is inversely related to growth rate (Thorpe 1977; Treasurer 1981) and, in a stunted population, males matured at an age of 2-3 years and females at 3-6 years, equating to a length of 70-90 mm TL. In a fast-growing, Western Australian population, males matured in their first year of life whereas the majority of females matured in their second year (Morgan et al. 2002). Females can produce 5,000-80,000 eggs depending on their size and these are all shed at the same time. Spawning occurs at night (Merrick and Schmida 1984) in late winter to early spring when water temperatures are 11-12ºC (Lake 1971; Cadwallader and Backhouse 1983). Lake (1967b) found that redfin perch started spawning in ponds when water temperatures exceeded 11.5°C. In Australian waters, redfin perch start spawning in spring (August or September) and the spawning season is over by the end of October (Weatherley and Lake 1967; Morgan et al. 2002). The eggs are strewn over aquatic plants and other submerged objects, such as wood debris, in long (up to 3 m), clear, gelatinous strings. The gelatinous coating is unpalatable to other fish so the eggs are protected from predation by fish (Froese and Pauly 2007). The eggs hatch in 1-3 weeks depending on water temperature (Pen and Potter 1992) and the fry aggregate and form schools soon after hatching. Merrick and Schmida (1984) reported a hatching time of 7-8 days at water temperatures of 14-19ºC.

Population size and structure: Shortly after redfin perch are introduced to a new pond, dam or lake, growth rate and population size increase rapidly and large fish (up to 2kg) often characterize new populations (Sportsfish Australia 2007). If natural predators are scarce, the population continues to increase and the growth rate declines, resulting in a large population of stunted fish (maximum size 0.3 kg). The larger fish in redfin perch populations are reported to be mostly female (McDowall 1996).

Uses: Redfin perch is valued for its fine white flesh and in Europe forms part of the commercial catch in many large lakes and reservoirs. It is also targeted by anglers and underpins valuable recreational fisheries in Europe and the UK. Its role in aquaculture has been limited in Europe but is growing. A small but significant aquaculture industry exists in the USA for the closely related yellow perch (*Perca flavescens*). Early attempts to farm redfin perch in Australia failed and its major use is now as a recreational fishing species targeted by freshwater anglers. Redfin perch are currently fished for by recreational anglers and form the basis for a small commercial fishery in western Victoria and South Australia (Kailola et al. 1993). A review of the value and economic status of this fishery is provided in chapter seven of this report.

Impacts overseas: Fishbase (Froese and Pauly 2007) provides a list of 11 countries where redfin perch have been introduced, and noted that adverse effects have been recorded in Ireland, China, and Australia the effects of redfin perch introductions were unknown in all other countries. Although Fishbase provides a useful global overview of fish species, its coverage of the literature is limited and recent studies may not have been incorporated into it. It therefore provides a useful global starting point. The reports in Fishbase indicated that in Xinjiang (China), the introduction of redfin perch had resulted in the disappearance of an endemic fish (*Asipiorhynchus laticeps*) in Lake Bositen (Kottelat and Whitten 1996). However, evidence for impacts in Ireland and Australia was either anecdotal or limited.

Fishbase did not report the results of recent studies in New Zealand, which have confirmed the likelihood of adverse impacts in some New Zealand lakes. For example, Closs et al. (2002) used removal studies to demonstrate an impact of perch on bullies

(*Gobiomorphus cotidianus*) in small South Island tarns. This builds on an earlier study by Griffiths (1976), which found that redfin perch fed heavily on common bullies (*G. cotidianus*). Both Rowe and Smith (2002) and Ludgate and Closs (2003) have reported relatively low catch rates (and hence abundances) of common bullies in lakes and ponds containing redfin perch compared with reference lakes. Furthermore, galaxiids (*Galaxias maculatus)* and smelt (*Retropinna retropinna*) were both absent in a northern New Zealand dune lake dominated by redfin perch even though they could readily access the lake (Rowe and Smith 2002).

A number of European studies have now demonstrated the key role that piscivorous redfin perch (and pike) play in the maintenance of lake water quality (e.g. see review by Mehner et al. 2002). Juvenile perch eat zooplankton and large, stunted populations of perch can therefore contribute to reduced water clarity (Romare et al. 1999). However, adult perch are piscivorous and fish manipulation studies have clearly demonstrated that a reduction in adult perch can lead to the proliferation of zooplanktivorous species such as roach and bream that reduce water clarity. Consequently, perch stocking has been successfully utilised in a number of lakes to suppress planktivorous fish. As this can increase impacts on small native fish the goals of maintaining water quality and reducing impacts on native fish are in conflict.

Introduction and impacts in Australia: Bayly and Williams (1973) noted that redfin perch were the first alien fish to be successfully introduced to Australia. Eleven fish were liberated into Tasmanian streams in 1862. Cadwallader and Backhouse (1983) reported that 10 fish, from another shipment, were subsequently introduced to Ballarat in 1868 and seven were introduced to Lake Wendouree in Victoria (Lake 1959; Roughley 1971). In 1888, redfin perch from this source were liberated into many New South Wales streams (Weatherley and Lake 1967; Clarke et al. 2000). Lintermans et al. (1990) detailed their introduction and spread in the ACT from 1888. Redfin perch were introduced into Western Australia in 1892 (Coy 1979; Hutchinson and Armstrong 1993) and by 1988 were well established in the south western region, well south of Perth (Lane and McComb 1988). In a recent survey, Morgan et al. (2002) found that redfin perch occurred in Western Australian rivers from the Swan to the Warren. Pen and Potter (1992) describe a patchy distribution for redfin perch throughout most of southern Australia and Welcomme (1988) noted that although redfin perch were present in parts of Western Australia, Southern Australia, New South Wales, Victoria and Tasmania, they were not present in Queensland or the Northern Territory.

In Western Australia, McKay (1977) reported that redfin perch caused a decline in the black minnow (*Galaxiella nigrostriata*; formerly *Galaxias nigrostriata*), but this species does not usually occur in habitats likely to contain perch so it may have been

Galaxiella munda (D. Morgan, pers. comm.). In eastern Australia Cadwallader (1978) found a negative association between the abundance of redfin perch and a number of native fish species in the Murray River (eastern Australia), noting that the native fish species did not recover until redfin perch declined (after invasion by carp). Cadwallader (1978) found an inverse relationship between the catch of redfin perch and that of native fish species (primarily silver perch, golden perch, Murray cod and bony bream) from the Murray River, and Cadwallader and Backhouse (1983) stated that redfin perch undoubtedly compete for food and space with both Murray cod and golden perch even though small redfin perch were sometimes eaten by Murray cod. They noted that redfin perch were implicated in the decline of Macquarie perch in Lake Eildon and that pygmy perches, rainbow fishes, and the western carp gudgeon (*Hypseleotris klunzingeri*) were all likely to be adversely affected by introductions of redfin perch because they occupy similar ecotones. Tangible evidence of an impact of redfin perch on western carp gudgeon was provided by Faragher and Lintermans (1997). They observed an increase in this species' abundance in Lake Burley Griffin (ACT) following the decline of redfin perch caused by a disease outbreak. Welcomme (1988) reported that redfin perch fed on small endemic fish and may have affected galaxiids, pygmy perch (*Nannoperca australis*), and golden perch (*Macquaria ambigua*) in some rivers. Fletcher (1986) noted that redfin considerably reduced the numbers of golden perch fry when the latter were stocked into clear water environments. Redfin perch were also thought to have affected the western pygmy perch (*N. vittata*) in the Murray River (in Western Australia) because its distribution in this river was fragmented and showed little overlap with redfin perch (Hutchinson 1991). As redfin perch are known to prey heavily on *N. vittata* (Pen and Potter 1992), predation by redfin perch is most likely to be the cause of their decline in this river, even though some predation by trout and agonistic behaviour by gambusia may also be involved (Morgan et al. 2002). Morgan et al. (2002) noted that *N. vittata*, the nightfish (*Bostockia porosa*) and the mud minnow (*Galaxiella munda*) all virtually disappeared from the Big Brook Dam reservoir shortly after the introduction and proliferation of redfin perch in this reservoir. Pen and Potter (1992) had previously found that redfin perch also prey on *B. porosa* so the decline of *N. vittata* and *B. porosa* was attributed to predation by redfin perch.

Wager and Jackson (1993) indicated that redfin perch predation had reduced Ewen's pygmy perch (*Nannoperca variegata*) and the Yarra pygmy perch (*N. obscura*), and that redfin perch were a threat to dwarf galaxias (*Galaxiella pusilla*), Macquarie perch (*Macquaria australasica*), and trout cod (*Maccullochella macquariensis*). Redfin perch predation has also been implicated in the decline of the purple-spotted gudgeon (*Mogurnda adspersa*), now thought to be extinct in the River Torrens in Adelaide and other St Vincent catchments such as the Onkaparinga (South Australian Museum 2007). However, the introduction and spread of gambusia and carp (*Cyprinus carpio*), as well as habitat changes, may have also contributed to its demise in these rivers. Lake and Hannon (2002) found that redfin perch reduced the habitat available to the flathead gudgeon (*Philypnodon grandiceps*) in the Onkaparinga River. In Tasmania, the decline of the swan galaxiid (*Galaxias fontanus*) in the lower reaches of many streams was attributed to its inability to co-exist with redfin perch and brown trout (Crook and Sanger 1998; 1999). Hardie et al. (2006) indicated that redfin perch was viewed as a pest species in Tasmania because of its impacts on *G. fontanus* and potential impacts on other galaxiid species should it spread.

McDowall (1980) indicated that the introduction of redfin perch to some reservoirs was responsible for the decline of rainbow trout, another introduced species of sports fish. Baxter et al. (1985) found that stocked rainbow trout fingerlings were heavily preyed on by redfin perch, as did Molony et al. (2004) in south eastern Australia. Molony et al. (2004) indicated that because redfin perch were very efficient predators of newly stocked rainbow trout fry, only the more expensive yearlings could be stocked into reservoirs where redfin perch are now present.

In their review of the impacts of alien fish species in Australia, Clarke et al. (2000) summarised the types of interactions associated with the impact of redfin perch on native fish in Australia. Negative interactions were noted for nine native fish species, seven involving predation, and three involving competition for food and/or habitat. However, Pen and Potter (1992) indicated that, even though redfin perch were known predators of some native fish, and were very abundant in the Collie River, they had not had a detectable impact on indigenous fish such as *N. vittata*, *B. porosa* and the western minnow, *Galaxias occidentalis*. This was attributed to the fact that, in this river, the spawning habitats of these species were relatively inaccessible to redfin perch.

Although it is apparent that redfin perch can reduce the densities of some native fish species, the presence of high turbidity and abundant in-stream cover can reduce this impact (Morison 1989, in Lintermans et al. 1990). Similarly, impacts on native fish in the more complex and open environments provided by rivers may be much less severe than in the closed, static-water environments provided by reservoirs, lakes, billabongs and farm ponds. Such ‘physical’ environmental factors may explain why the severity of redfin perch impacts on native fish can vary between locations.

Redfin perch have indirectly affected Macquarie perch in ACT through the introduction and spread of the fish pathogen (EHNV) (Lintermans 1991). Ironically, redfin perch numbers ‘crashed’ in Lake Mulwala following the introduction of common carp (*Cyprinus carpio*) to this lake (NFA 2007). This is believed to be because carp reduce the survival of redfin perch eggs (loc. cit.), however, the high turbidity resulting from carp may also affect the feeding, foods and spawning habitat (i.e. macrophytes) of perch. The growth rates of Murray cod and other endemic species increased following these changes in the introduced ichthyofauna of Lake Mulwala but, in other waters, carp removal may result in an increase in redfin perch abundance and so create different problems for native species.

Redfin perch may also pose a threat to crayfish populations in Australia. Pen and Potter (1992), Beatty (2000), Morgan et al. (2002), and Morgan et al. (2004) have all noted the propensity of larger redfin perch to prey on marron (*Cherax cainii*) and hence to reduce marron populations in Western Australian waters.

Overall, there was some evidence that redfin perch have adversely affected the abundance and/or distribution of 14 native fish species (Table 4.3). Most of this evidence was circumstantial and based on field studies showing that there was either a decline in the abundance of a native fish species following the introduction of redfin perch or that the native species was much less abundant in waters containing redfin perch compared to waters lacking redfin perch.

Table 4.3 Classification of studies on the impact of redfin perch (*Perca fluviatilis*) on native fish in Australia.

No.	Common name	Scientific name	Studies providing evidence of impact (numbers refer to studies listed below)	
			Change in abundance	Perch predation
1	Golden perch	*Macquaria ambigua*	3	2
2	Purple-spotted gudgeon	*Mogurnda adspersa*	13	
3	Western carp gudgeon	*Hypseleotris klunzingeri*	8	
4	Midgley's carp gudgeon	*Hypseleotris sp. 1*	12	
5	Western pygmy perch	*Nannoperca vittata*	5,9	6
6	Yarra pygmy perch	*Nannoperca obscura*	7	
7	Nightfish	*Bostockia porosa*	9	6
8	Ewen's pygmy perch	*Nannoperca variegata*	7	
9	Southern pygmy perch	*Nannoperca australis*	3	
10	Flat-head galaxias	*Galaxias rostratus*	11	
11	Mud minnow	*Galaxiella munda*	9	
12	Western minnow	*Galaxias occidentalis*	1	
13	Flathead gudgeon	*Philypnodon grandiceps*	10	
14	Murray cod	*Maccullochella peelii*	4	

[1]Beatty and Morgan (2005) and D. Morgan (unpublished data), [2]Fletcher (1986), [3]Welcome (1988), [4]Lintermans et al. (1990), [5]Hutchinson (1991), [6]Pen and Potter (1992), [7]Wager and Jackson (1993), [8]Faragher and Lintermans (1997), [9]Morgan et al. (2002, 2004), [10]Lake and Hannan (2002), [11]McNeil (2004) cited in Wilson (2005), [12]Stoffels and Humphries (2003), [13]South Australian Museum (2007),

The main mechanism identified to date is predation, and most evidence for this is provided for *N. vittata*. Few studies addressed the possibility of reduced or changed fish distributions caused by either an increased predation risk or competition for food, and there have been no controlled manipulation studies to confirm the impact of redfin perch alone. The increased abundance of *H. klunzingeri* in Lake Burley Griffin following a high natural mortality of redfin perch (Faragher and Lintermans 1997) is akin to a natural manipulation experiment but data on the pre-perch abundance of *H. klunzingeri* in this lake are required to confirm this. The removal of redfin perch from Phillips Creek reservoir in Western Australia (through draining) resulted in approximately 100,000 *G. occidentalis* colonising the dam despite this species not being previously recorded when redfin perch were present (Beatty and Morgan 2005, D.L. Morgan unpublished data). This result strongly suggests an impact of redfin perch on *G. occidentalis*, but other coincidental factors that could also account for an increase in native fish need to be excluded to confirm the role of redfin perch.

4.3 Tench (*Tinca tinca*)

Description, subspecies, and hybridisation: With a maximum length of 70 cm, adult tench are relatively large fish. They are olive-green to dark bronze in colour, and are found mainly in still or slow-moving freshwater environments. Tench have large soft fin rays, two barbels, and red eyes, which together with their small scales are their most distinctive features. Adult tench are readily distinguished from all other species by these features and a key for the identification of this species in Australian waters is provided by McDowall (1996). Tench are renowned for their slime production and are also known as the ‘doctor’ fish because of the reputed therapeutic action of this mucous layer.

Sexual dimorphism is often apparent, with males having larger pelvic fins than females. Males also have enlarged second fin rays and a muscular protuberance extending from the flank (Vainikka 2003; Coad 2003). However, Muus et al. (1967) indicated that such dimorphism is only apparent after age 2. A further potentially complicating factor in identification of the sexes is that triploid fish may occur naturally in some populations and have intermediate-sized fins. Weatherley (1959) could not reliably identify the sex of fish less than 10-12 cm long in Tasmanian waters on the basis of external appearance.

Although tench can be crossed with a range of other cyprinids, including goldfish, common carp, rudd and orfe (Victorovsky 1966; BISON 2003), there is no primary evidence for hybridisation in the wild. Colour variants (e.g., golden tench) occur overseas and may represent the results of such hybridisation.

Indigenous range: Tench are a European fish and their natural range is likely to have extended throughout northern Europe encompassing all the rivers of the Baltic, Caspian and Black Sea and extending westward to the Ob and Yenisei River basins in Siberia (Berg 1949). Historical introductions have probably led to an expansion of the natural range further west to Portugal and parts of Spain, south to Greece and parts of

Italy, north to Norway and Finland, and east to India. As tench were introduced from England to Ireland in the 18th century, it is possible that they were introduced to England at an earlier date. Froese and Pauly (2007) reports a latitudinal range of 64ºN to 36ºN, and a longitudinal range of 10ºW to 104ºE.

Habitats and migrations: Rendon et al. (2003) examined the substrate preferences exhibited by tench and found a clear preference for mud over other substrates including sand, artificial vegetation, and concrete. Adult tench inhabit a range of waters characterised by low water velocity, soft substrates (e.g., mud, silt or sand), and the presence of some aquatic vegetation. Such habitats include the lower reaches of rivers, off-river habitats such as oxbows and river deltas, the shallow margins of lakes, drainage canals, estuarine areas, wetlands, and shipping canals (Bouvet et al. 1984; Townsend and Peirson 1988; Rossier 1995; Pilcher and Copp 1997; Donnelly et al. 1998; Gonzalez et al. 2000; Coad 2003). In general, tench inhabit the shallower regions of these habitats. Froese and Pauly (2007) reported a depth range of 1 m, however, large (20-30 cm long) tench have been captured in nets at depths of 7-15 m in several New Zealand lakes (unpubl. data).

Tench are found mainly in large rivers/streams with mean flows over 28 m^3/s (BISON 2003), presumably because in such rivers the lower reaches contain large areas of low water velocity habitat. In lakes, they prefer shallow, near-shore habitats (Rossier 1995), presumably because of the presence of macrophytes. In a study of 53 Mazurian lakes over 40 ha in area, Szajnowski (1970) found a strong relationship between the number of tench caught and the ratio of littoral zone area to total lake surface area. The micro-habitat of juvenile tench is shallow water with a silty bottom, such as the dense, millfoil and pondweed filled off-channel sites found in many English rivers (Copp 1997).

Tench are generally benthic and forage for food mainly at night, covering a wide search area (Perrow et al. 1996). However, Vainikka (2003) indicated that on calm, sunny days some fish fed on terrestrial insects at the water surface. Perrow et al. (1996) followed adult tench using radio telemetry and found that they were inactive during the day and rested in preferred locations associated within the rush *Typha*. This was thought to be because *Typha* has relatively wide stems that permit access by tench to deep cover. Radio-telemetry studies also indicated that they were relatively sedentary in a side arm of the Rhone River (Bouvet et al. 1984), but that they migrated to and from specific locations in a shipping canal (Donnelly et al. 1998). Weatherley (1959) indicated that tench in Tasmanian waters aggregated in deep holes and shady areas. They can also be expected to over-winter in such locations. Coad (2003) reported that, in winter, tench in Iran were largely inactive and buried themselves in shallow muddy habitats. Such behaviour is likely to occur in severe winters (Wheeler

1969). In a UK river, they preferred the downstream, channelised sections during winter months (Pilcher and Copp 1997).

Because of their ability to tolerate low oxygen levels (see below), tench can inhabit the deeper, hypolimnetic zones of lakes and ponds when oxygen levels are low and these habitats cannot be utilised by most other fish species (BISON 2003).

Tolerances and limiting factors: Tench are often referred to as a warm-water fish and, unlike cold-water salmonids, they prefer temperatures over 20ºC. Their preferred temperature is 20-21ºC and they have a final preferendum of 27.4 ± 0.5ºC (Perez Regadera et al. 1994). Coad (2003) reported a preferred range of 15 to 23.5ºC and in tanks providing a temperature gradient, tench inhabited waters between 20 to 24ºC, rarely venturing into waters over 25ºC (Alabaster and Downing 1966). However, tench have been reported to tolerate waters up to 37ºC for brief periods (Coad 2003). Weatherley (1959) carried out tank experiments on the lethal temperature for tench in Tasmania and found that it increased from 27.5ºC to 33.5ºC depending on the time of year and hence to the acclimation temperature of the tench. BISON (2003) indicated an upper lethal temperature of 35.2ºC. Juvenile tench may require somewhat warmer water temperatures than adults. For example, Hamackova et al. (1995) found that a temperature reduction below 22ºC increased the mortality rate of 2-4 day old fry, but not 7-10 day old fry. The extent of mortality was directly related to the size of the temperature drop.

Tench are highly tolerant of low oxygen levels (Vainikka 2003; Coad 2003) and can survive in waters where oxygen levels are as low as 0.7 mg/l (BISON 2003).

Adult tench are tolerant to a range of pHs variation and prefer the range 6.5-8.0 (BISON 2003). Mortality increases at pH below 5 and above 10.8. Similar ranges were reported for larvae. For example, Hamackova et al. (1998) found that larval survival was highest in the pH range 7-9, but some survival occurred at a pH of 5 as well as 10. Values of 4 and 11 were lethal to all larvae.

Tench are also tolerant of moderately brackish water. Weatherley (1959) found that although a salinity of 15.4 ppt was fatal within 24 h, tench were able to withstand 13.8 ppt, albeit with greatly reduced motor functions. Coad (2003) reported a tolerance to 12 ppt salinity. Tench can thrive in brackish waters such as estuaries and the Baltic Sea, where salinities can range from 4-10 ppt (Weatherley 1959).

Tench prefer low water velocities and avoid the high-gradient reaches of rivers and streams where water velocities are high. BISON (2003) indicated a maximum water velocity of 0.27 m/s. Tench also prefer shallow waters: BISON (2003) indicated a maximum depth of 7.6 m, but this is likely to represent the maximum depth at which tench have been captured rather than a maximum depth tolerated.

Tench are reported to be nocturnal; however, this is probably related to predator avoidance rather than intolerance of light. Garcia-Ceballos et al. (1998) found that tench under low light (40 lux) were gregarious, but this behaviour declined as light levels increased up to 200 lux. Tench thrive in both clear and turbid waters, so the high suspended solids levels occurring in turbid lakes are unlikely to affect them.

San Juan (1995) listed the main factors known to limit the size of tench populations. These were water level fluctuations, increased exposure to wave action, destruction of fry habitat, and effects of predators. Wolter et al. (2000) indicated that loss of macrophytes could also be a factor that limits tench populations in lakes, and Hinrichs (1998) noted the adverse impact on tench of drain maintenance activities, such as removal of vegetation. The spawning and recruitment of tench in Lake Parkinson, New Zealand, was not suppressed by removal of all aquatic vegetation; tench remained the most abundant species in this lake two years after total weed removal had occurred (Rowe and Champion 1994; Rowe 2004).

Predators, parasites and diseases: The main aquatic predator of tench in European waters is the pike (*Esox lucius*). However, tench are also vulnerable to predation by large-mouthed, or black bass (*Micropterus salmoides*) (BISON 2003; Garcia-Berthou and Moreno-Amich 2000). Bronmark et al. (1995) determined the role of piscivores in a range of Swedish lakes. They found that tench populations in lakes with piscivorous fish were characterised by low population size and a prevalence of large fish. In lakes lacking piscivores, tench populations were large and composed mainly of small fish.

The main parasites of tench have been described by Yildiz (2003) and Ozturk (2002). Helminthes are the main endoparasites and include species such as *Asymphylodora tincae*, *Pomphorhynchus laevis*, and *Acanthocephalis lucii* that occur in the intestine; *Ligula intestinalis* and *Pomphorhynchus laevis* that occur in the abdominal cavity; nematode larvae, metacercariae and *Piscicola geometra* that are found in the skin; and *Dactylogyrus macracanthus*, *Argulus foliaceus* and *Ergasilius sieboldi* that occur on the gills. Yildiz et al. (2003) indicated that infection rates for *Ligula* ranged from 41-84 % and declined with fish size.

Age, growth and size: Maximum length is reported in Froese and Pauly (2007) to be 84 cm TL and Lake (1967c) reported a maximum weight of 9 kg for tench in Australia. The maximum age is estimated to be 36 years (Froese and Pauly 2007).

Tench in Tasmanian ponds grew at maximum rates of 2 cm/month over their first thirteen months, but growth then slowed such that over four years growth rate averaged 7 cm/year (Weatherly 1959). This was the fastest growth rate recorded and tench in Lake Tiberias (Tasmania) grew at less than half this rate (i.e., 10 cm by age four). The relatively slow growth rate of tench in Tasmanian rivers and lakes was not thought to be related to water temperature (Weatherley and Lake 1967), but growth rates were expected to be higher in the warmer waters of New South Wales. For example, Weatherley and Lake (1967) indicated that the largest tench in Tasmania were rarely over 1 kg compared with 2 kg in New South Wales. As growth rates for tench in New South Wales were not determined, the larger size of New South Wales tench could simply be related to a greater longevity.

Sixty, 80 mm long tench were stocked into Lake Parkinson (New Zealand) in September 1974 to establish a recreational fishery. By 1976, a sample of 225 tench revealed fish ranging in size from 4 to 39 cm (Fig. 4.2). The original tench had grown from a mean length of 8 cm in 1974 to over 30 cm by 1976, indicating an average annual increment of approximately 11 cm/yr (Rowe 2004). This can be expected to be a relatively fast growth rate as the population density for these 60 fish was low.

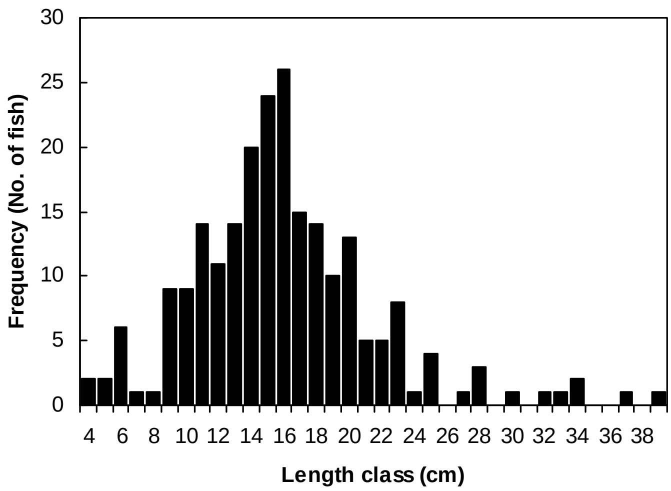


Figure 4.2 Population size-structure for tench in a small (2 ha), shallow (7 m max. depth) and weedy lake near Auckland, New Zealand (from Rowe 2004).

Feeding and diet: Tench are generally bottom dwelling and are thought to use taste and olfactory cues to locate animal prey. Studies of taste reception to improve food palatability for aquaculture purposes indicated a strong preference for the amino acids cysteine and maleic acid (Kasumyan and Prokopova 2001). Several studies have indicated that tench are nocturnal foragers (Herrero et al. 2003; Perrow et al. 1996), so visual cues may be less important than taste and olfactory cues for prey location. This would allow tench to thrive in turbid as well as clear waters.

Tench have no teeth, but have a relatively large mouth, and probably feed using suction to ingest their prey. However, Petridis (1990) also observed tench to feed by using their buccal cavity to squirt water at the surface layer of silt that overlies lake beds. This suspends small interstitial prey such as chironomids, micro-crustacea and oligochaetes in a small cloud of silt, and the tench then ingest these.

Studies of the diet of tench indicate that they feed primarily on benthic macro-invertebrates, although a number of authors also report feeding on zooplankton and adult Insecta, indicating mid-water and surface feeding, respectively (Weatherley 1959; Ranta and Nuutinen 1984; Giles et al. 1990; Michel and Oberdorff 1995; Perez-Bote et al. 1998; Gonzalez et al. 2000). Food items recorded for tench include zooplankton (cladocerans, copepods, and ostracods), benthic crustacea (amphipods and decapods), benthic insecta (chironomids, Odonata, Ephemeroptera, Hemiptera, Corixidae, and Hirudinea), and bivalves (Gastropoda and small Bivalvia). They are therefore capable of preying on most aquatic invertebrates. Larger fish can utilise very small prey and are therefore not dependent on relatively large prey for fast growth, as is the case for some freshwater fish. Giles et al. (1990) found that diet varied greatly between two UK gravel pit ponds, from mainly *Daphnia* in one to mainly benthic invertebrates, including bivalves and crustaceans, in the other. Gonzalez et al. (2000) reported heavy feeding on chironomid larvae in both riverine and lacustrine environments, with crustacea the second most important food in the lake and gastropods in the river. Few studies of prey selectivity have been carried out. However, Petridis (1990) recorded positive selection for the isopod *Asellus aquaticus* in a reach of the Lancaster Canal, despite the higher abundance of gastropods and chironomids. Negative selection for chironomids occurred here.

BISON (2003) indicated that tench larvae will feed on algae, phytoplankton (including blue-green species), zooplankton, rotifers, and water mites. Ranta and Nuutinen (1984) found that small tench preferred large *Daphnia* (1.5-4 mm long) and eliminated these before turning to smaller planktonic prey. Pyka (1997) reported a daily food ration for juvenile tench (weight 44 mg) of about 6.6 % at 25ºC.

Overall, tench can be regarded as a generalised, benthic, carnivore (i.e., a benthophagous species), with the predominant prey being those that are most readily available. Large, soft-bodied Crustacea are probably preferred over smaller prey that are more difficult to obtain (e.g., oligochaetes and chironomids), or hard-bodied prey (e.g., molluscs). Some reports indicate that tench also feed on algae and macrophytes, however, this is thought to be rare and to occur by mistake, or when benthic invertebrates are scarce (Weatherley 1959; Coad 2003; Wheeler 1969).

There are few studies on the diet of tench in Southern Hemisphere waters. Rowe (2004) found that the main prey species in a small New Zealand lake (by percent occurrence) was the cladoceran *Bosmina* (62.2 % in 1976 and 61.1 % in 1977). *Bosmina* were often packed into the stomachs of quite large (FL > 400 mm) fish, and it is unlikely that these small (<1 mm diameter) prey were taken individually by such large fish. Large tench may therefore be able to filter small prey items from the sediment surface. Petridis (1990) observed tench feeding and noted that they could feed on the very small animals present on and within the surface layer of silt by suspending them in a cloud of water and then extracting them from this.

In Lake Tiberias (Tasmania), Weatherley (1959) found that young tench fed mainly on zooplankton with amphipods and insect larvae being increasingly utilised by the larger (> 100 mm) fish. In three other Tasmanian waters, adult tench fed mainly on pulmonate molluscs, oligochaetes and chironomid larvae, respectively.

Maturation, spawning and fecundity: Tench can mature at 2+ years old at a size of 90 g (males) and 110 g (females) (Sanchez-Herrera et al. 1997). However, age at first maturation is likely to vary between locations depending on growth rate and water temperature. Neophitou (1993) reported that both sexes of tench in a Greek population matured first at age 3+. Yilmaz (2002) reported maturation ages of 3-4, and that in the colder and more northern waters of Finland, tench don't become sexually mature until they are 4 or 5 years old (Vainikka 2003). In experiments designed to determine the effect of temperature on tench reproduction, Horoszewicz et al. (1977) found that females in a warm pond matured earlier, had higher fecundities, and spawned more often than tench in a colder pond.

The seasonal timing of tench spawning appears to be controlled by both water temperature and photoperiod. Shikhshabekov (1977) and Neophitou (1993) reported that spawning in Dagestan and Greece respectively occurred in spring when water temperatures ranged from 18-20ºC. For at least some populations, the date of first spawning can be predicted by determining the sum of degree-days over 10ºC (Breton et al. 1980; Horoszewicz 1983). However, photoperiod can modify the influence of water temperature on the timing of spawning (Martin et al. 1999). For example, Weatherley and Lake (1967) found that in Australia, tench spawn in summer (December to February), rather than in spring.

Spawning occurs in shallow (usually <1 m deep) waters, and tench are broadcast spawners, scattering their eggs over aquatic vegetation such as macrophytes and reeds. The eggs stick to the vegetation and are small (0.9-1.0 mm in diameter) and green coloured. Groups of males have been observed following one or more female prior to spawning. The males are believed to be attracted to the females through pheromones released into the water via the gills. Pinillos et al. (2002) found that male sensory systems were strongly activated by free and glucorinated prostaglandins. Tench require aquatic plants, wood debris, or some hard substrate for successful recruitment.

They have been observed to spawn on the bare mud bottom of ponds, but few fish resulted and they are not able to spawn successfully in Tasmanian farm dams (Weatherley and Lake 1967), probably because of the absence of any substrate other than silt.

Female fecundity is relatively high. Pimpicka (1991) reported fecundities of 85,700-543,900 eggs per kg of fish and Neophitou (1993) recorded an average of 184,000 ± 21,200 eggs per kg for a tench population in Greece. Froese and Pauly (2007) reported a maximum fecundity of 800,000 eggs. However, not all eggs are spawned at once. Tench are batch spawners and 3-9 spawnings may occur over the spawning season (Alas & Solak 2004). Furthermore, not all females participate in each spawning event (Horoszewicz 1983). In colder climates, there may be fewer spawnings per season, with Shikhshabekov (1977) reporting only two for tench in Dagestan.

Incubation of eggs occurs in 76 hours at a mean water temperature of 19.6 ºC (Penaz et al. 1981), and the highest incubation rate (89.4 %) occurred at 22.9 ºC with fry hatching after 48 hours (Kouril et al. 1988). On hatching, larvae are about 3.8 mm long (Penaz et al. 1981). Tench larvae have an attachment organ, which allows them to latch onto the under-surfaces of plants (Coad 2003). Exogenous feeding occurred after 11 days, at a length of 5.6 mm (Penaz et al. 1981) and tench larvae can be expected to be free-swimming beyond this size.

Population size and structure: The population density of tench varies between environments, and has been found to range from 126 to 530 individuals/ha (Lusk et al. 1998; Wright and Giles 1991). Lusk et al. (1998) reported a mean of 367 individuals/ha and a mean biomass of 123.5 kg/ha (range 12.4-260.5). In mixed species communities of fish, tench accounted for 8 % of all fish by numbers and about 25 % of all fish by biomass. Wright and Giles (1991) reported a standing crop of 102 kg/ha in St Peters Lake (UK), which had abundant vegetation, but only 0.5 kg/ha in the more turbid, weed-free Main Lake. In St Peter's Lake, tench accounted for 29 % of total fish biomass. Ziliukiene (1993) indicated that the production of tench could be up to 4.3 kg/ha. Wright and Giles (1991) indicated that strong year classes occurred during warm summers, with year class strength being positively correlated with degree days over 16ºC (r = 0.58).

When Lake Parkinson, New Zealand, was rotenoned in 1981, a fish census was carried out and the total number of all tench present at that time was 3,560. This gives an areal density of 1874 fish/ha (Rowe and Champion 1994). This higher than expected density was probably because all young-of-the-year fish were included in the census and this is usually not possible with more conventional sampling methods, which often miss most juveniles. The total biomass of tench was 72.7 kg or 36.4 kg/ha.

The sex ratio of tench in Lake Parkinson was strongly skewed towards females. In a sample of 963 fish ranging in length from 75-535 mm, 62.6 % were females. Females were more numerous than males for all size classes of fish, and all fish over 400 mm long were females. The sex ratio of tench in New Mexico was reported as 1:1 (BISON 2003). In Finland it is thought to be close to 1:3 (males: females) because of a high mortality of smaller males (Vainikka 2003). Data presented by Wright and Giles (1991) also indicate a prevalence of females (56 %) even though the growth rates, and hence sizes of males and females, were very similar.

Uses: The tench (*Tinca tinca* L.*)* is still an important, albeit small, component of commercial catches in many large European lakes (Ziliukiene and Ziliukas 1998; Grosch et al. 2000). Today, interest in the aquaculture of tench is increasing rapidly in southern Europe (Billard et al. 1995; Reader 1998) and the aquaculture of this species may soon replace the harvest of wild fish as the main commercial source. The tench is also a major sports fish in Europe and forms an important part of the 'coarse fish' recreational fishery in both Europe and the UK.

Impacts overseas: The occurrence of ecological impacts from the introduction of tench to waters beyond their indigenous range was determined for seven of the 26 countries where introductions had occurred (Froese and Pauly 2007). No effects were listed for the 18th century introduction from England to Ireland but it is doubtful whether these would have been recognised at that time. More recently, both Kennedy and Fitzmaurice (1970) and O'Maoileidigh and Bracken (1989) reported on the biology of tench in Ireland and made no mention of adverse effects. Adverse impacts were stated as 'unknown' for the introductions to USA, Finland, and Tunisia and 'probably no' for New Zealand and Portugal. However, Baughman (1947) reported that tench were regarded as a nuisance in parts of Maryland and Idaho because of their high abundance. Perez et al. (2003) indicated that a number of alien fish species introduced to Chile for aquaculture purposes, including tench, had created problems for native fish.

Giles et al. (1990) indicated that, on the basis of their diet and food preferences, trophic overlap could occur between tench, redfin perch and wildfowl in shallow waters. Trophic overlap was also expected between tench and common bullies (*Gobiomorphus cotidianus*) in Lake Parkinson, New Zealand, because both species were benthic omnivores, and abundant in this lake. However, there was no evidence for an impact of tench on the common bully (Rowe 2004). The density and size of common bullies was relatively high in this lake (Mitchell 1986) despite the presence of a high population of tench (Rowe and Champion 1994).

Direct effects of tench on other fish appear to be negligible, but other fish may affect tench. Tench have been reduced by introductions of some alien fish species, including piscivores such as large-mouthed bass (Garcia-Berthou and Moreno-Amich 2000). Conversely, tench populations were apparently enhanced by eel stocking (Leopold 1986).

Although reports of tench interactions with other fish species are rare, the tench has been implicated in reduced densities of some invertebrates. In enclosures, tench reduced gastropods, but not other macro-invertebrates (Beklioglu and Moss 1998; Bronmark 1994).

Tench are also known to be selective planktivores and so may exert top-down effects (i.e., a reduction in zooplankton) on some lake ecosystems, thereby increasing phytoplankton and reducing water clarity. Ranta and Nuutinen (1984) demonstrated strong selection by tench for large *Daphnia*, and Perez-Bote and Limpo-Iglesia (1998) found that the zooplankton composition of tench ponds varied seasonally, with *Daphnia* and copepods dominating in winter when tench are inactive, but not in summer when tench were active. Small rotifers dominated the plankton in summer months. Beklioglu et al. (2003) carried out a partial removal experiment in a lake dominated by tench. The tench in this lake fed mainly on zooplankton and a 250 % improvement in water clarity followed removal of 57 % of the total fish stock. This improvement in water clarity was attributed primarily to the top-down role of tench, but common carp were also present in this lake, so they may have also contributed to its turbidity. There was no direct evidence for an effect of tench on water clarity in Lake Parkinson, New Zealand; however, water clarity did improve immediately after removal of all fish including tench (Rowe and Champion 1994). As tench were the dominant species, and their main food was the cladoceran, *Bosmina*, it seems likely that they will have contributed to the reduction in water clarity caused by planktivores in Lake Parkinson.

Tench may also change lake ecosystems through bottom-up effects on food webs. A number of studies have shown that tench can reduce macrophyte production by stimulating greater periphyton growth on macrophyte surfaces (Bronmark 1994; Beklioglu and Moss 1998; Williams et al. 2002). The increased periphyton cover reduces light penetration and nutrient supply to macrophytes, resulting in their decline. The increase in periphyton may be related to removal of browsing gastropods by tench (Bronmark 1994; Beklioglu and Moss 1998) and/or to stimulation of periphyton growth through the increased cycling of inorganic nitrogen through tench excreta (Williams et al. 2002). Phosphorus concentrations were high in all treatments, so were not implicated (Williams et al. 2002). Such effects have only been recorded when densities of tench were relatively high, and Williams et al. (2000) indicated that a tench biomass in excess of 200 kg/ha may be required for macrophyte reduction.

Tench may also increase turbidity in lakes through their foraging activities, but again this would only be expected at high densities when food becomes scarce, and tench are forced to forage in surficial sediments to find small prey. Tench have been observed feeding by 'squirting' water at the surface layer of sediment to suspend the overlying silt (Petridis 1990). Tench in South Africa caused an increase in the turbidity of shallow waters by disturbing bottom sediments (de Moor and Bruton 1988).

In summary, tench interactions with other fish appear to be minor, but at high densities they may have an adverse effect on macrophytes and water quality, and hence on habitat for some fish species.

Introduction and impacts in Australia: According to McKay (1984) tench were first introduced from England to Australia through importations to Tasmania in the 1870s. However, Clements (1988) noted an earlier introduction, to a reservoir near Cascade in Tasmania, in the late 1850s. Another introduction, from England to Melbourne occurred in the early 1960's (Clements 1988). In 1876, they were introduced to a number of Victorian waters and to the River Murray from where they spread to the middle and lower reaches of Murray-Darling River system (McKay 1984). By 1886, they had been spread to New South Wales (Weatherley and Lake 1967). They are now also present in South Australia (Cadwallader and Backhouse 1983). Tench were released into Western Australia (near Albany, York and Perth) in the 1890s (Coy 1979), but are no longer likely to be present there. Their current distribution within Australia is shown in Chapter 2.

Some anecdotal information on population size is provided by Brumley (1991). She reported a relatively high abundance of tench in some parts of the Murray-Darling system (west of the divide) and a small population also occurred in the Onkaparinga River but declined drastically after the introduction of carp (South Australian DEH 2007). Tench formed a small commercial fishery in South Australian waters until they declined following the spread of carp and the expected competition for food with carp (Reynolds 1976; 1979 in Brumley 1991). Weatherley and Lake (1967) indicated that, in the rivers where they occurred, tench were only abundant in slow-flowing waters with weedy substrates. This observation suggests that tench will occur primarily in the lower, slow-moving and macrophyte-dominated reaches of rivers. Today, the abundance of tench is likely to be highest in the Derwent River in Tasmania (Brumley 1996)

Cadwallader and Backhouse (1983) reported that nothing was known about the impact of tench on the native fauna of Australia. Later, Arthington and Blühdorn (1995) noted that both P. Gehrke and P.S. Lake thought that impacts of tench in Victorian and New South Wales waters would be small because tench abundance was low. Arthington and McKenzie (1997) observed that there was relatively little information on the impacts of tench on either Australian native fish or aquatic ecosystems, and that this situation has not changed over the past decade.

McDowall (1996) indicated that tench were once commercially fished for in the Murray River but the fishery had declined as common carp became abundant in the 1970s and reduced the abundance of tench. Cadwallader and Backhouse (1983) noted that tench were once used as crayfish bait in South Australian waters, but this practice declined following the proliferation of common carp there.

The only recorded impact of tench in Australian waters is a decline in water clarity. Merrick and Schmida (1984) proposed that the feeding habits of tench were the cause of muddy waters in dams and ponds and they cited comments by Lake (1967c), Weatherley and Lake (1967), Maclean (1975), Lake and Bennison (1977) and Mitchell (1979) in support of this.

4.4 Roach (*Rutilus rutilus*)

Description, subspecies, and hybridisation: Cadwallader and Backhouse (1983) described roach as a small, laterally compressed fish with a relatively deep-body. It has an arched back, a strongly forked tail and a small mouth. It is silvery coloured (darker above and lighter below) with bright red eyes. Roach have a single dorsal fin with a relatively square edge and red pectoral and pelvic fins, and they possess small auxiliary processes at the base of the pelvic fins. The scales are moderately sized and cycloid. A key for the identification of this species in Australian waters is provided by McDowall (1996).

Froese and Pauly (2007) report the existence of at least 6 subspecies: *Rutilus rutilus caspicus* from the Caspian Sea; *Rutilus rutilus heckeli*, *Rutilus rutilus fluviatilis*, *Rutilus rutilus aralensis* and *Rutilis rutilus schelkovnikovi* from the Aras River basin; and *Rutilus rutilus uzboicus* from the Uzboi Valley in Turkmenistan. Roach readily hybridise with bream and hybrids are common in areas where the two species co-exist (http:/fishing.agrino.org).

Indigenous range: Originally, roach were widespread in Europe, being absent only from Spain, Italy, Greece and Ireland (Froese and Pauly 2007). They are reported from a latitudinal range of 71ºN to 36ºN and a longitudinal range of 10ºW to 155ºE.

Tolerances and limiting factors: In Australia, roach are found where water temperatures range from 8-25ºC but they can tolerate temperatures from 0ºC up to 38ºC (Cadwallader and Backhouse 1983). Froese and Pauly (2007) report a typical temperature range of 10-20ºC, a pH range of 7-7.5, and some tolerance of brackish water. Roach are common in the north Baltic Sea where salinities range from 5.5-6.5 ppt (Lappalainen et al. 2005). Roach can also thrive in poor quality and polluted waters (Froese and Pauly 2007).

Habitats and migrations: The roach is a schooling fish and, in Europe, it is a benthopelagic species found in slow-flowing or still, muddy waters (Froese and Pauly 2007). Roach therefore inhabit rivers, lakes, canals and reservoirs. In Europe, they are thought to prefer shallow waters, 2-3 m deep, with rocky, weedy or sandy substrates (http:/fishing.agrino.org). Feeding occurs throughout the day, but roach are more active at dawn and dusk than at other times. In the Kalavasos Dam, predation by bass resulted in a shift in the distribution of smaller fish away from shallow littoral areas to deeper waters near the centre of the reservoir, with feeding then occurring in the littoral zone at night (http:/fishing.agrino.org). However, roach in a Norwegian lake fed on zooplankton in the limnetic zone at night and on the benthos in the littoral zone during the day (Braband et al. 1984). Such diurnal and spatial changes in feeding reflect the plasticity in habitat use by roach. Roach inhabit the brackish waters of the Baltic and Black sea where anadromous populations occur. Lacustrine roach migrate to spawning grounds in tributary streams (L'Abee-Lund and Vollestad 1985). In Australia, Clements (1998) observed large spawning migrations of roach in the Burumbeet Creek, and both up and downstream movements have been recorded in riverine populations in Victoria (Merrick and Schmida 1984).

Predators, parasites and diseases: Natural predators in Europe include piscivorous fish such as the eel (*Anguilla anguilla*), brown trout (*Salmo trutta*), pike (*Esox lucius*), shad (*Alosa macedonaica*), dace (*Leuciscus cephalus*), burbot (*Lota lota*), wels (*Siluris glanis*) and the pike-perch (*Sander lucioperca*) (Froese and Pauly 2007). Redfin perch (*Perca fluviatilus*) are also a significant predator of roach in some European waters (http:/fishing.agrino.org). Avian predators such as cormorants (*Phalacrocorax* sp.) can also be expected to prey on roach. In Australia, roach are thought to be a forage fish for Murray cod and the golden perch (Cadwallader and Backhouse 1983).

Roach are vulnerable to some parasites. During 1975, extensive mortalities occurred in Slapton Ley (Devon, UK) as a result of in infestation by *Ligula intestinalis* (Burrough and Kennedy 1979). This species of tapeworm is present in Australia.

Age, growth and size: Merrick and Schmida (1984) report a maximum length of 450 mm but more usually 150-200 mm. This agrees well with the maximum length of 460 mm reported in Froese and Pauly (2007), and the maximum theoretical size estimated from Ford-Walford plots for female roach in the River Frome (Mann 1973). The growth rate of roach differs between the sexes, being faster for females than for males (Mann 1973). The maximum reported age in European populations is 18 years (Holcik

1967; Mann 1973) but Backhouse and Cadwallader (1983) reported a maximum age of 12 years for roach in Australia. Mann (1973) reported a number of studies indicating that high water temperature in summer favoured fast growth by roach. However, roach have an optimal temperature for growth and in a number of European populations, the growth of female roach decreased with increasing latitudinal distance away from the centre of this species' geographical distribution (Lappalainen et al. 2006). This is thought to reflect the effect of colder and warmer water temperatures, respectively, on growth.

Feeding and diet: Roach have no true teeth and feed mainly on zooplankton as juveniles, but in Australia adults are primarily benthic omnivores (Cadwallader and Backhouse 1983). Roach possess pharyngeal teeth (McDowall 1996) so they have some capacity for consuming plants, as well as small benthic organisms. Froese and Pauly (2007) reported a diet of insects, crustaceans, molluscs and plants, with juveniles being primarily planktivorous and adults preferring to feed on plants. Feeding occurs during the day, but peaks in feeding activity occur at dawn and dusk. In a study of roach in two UK gravel pits, roach fed primarily on filamentous algae in one and on water fleas (*Daphnia* sp) in the other (Giles et al. 1990). Mann (1973) reported that roach generally ate chironomid larvae but fed heavily on molluscs at certain times of the year. In a Norwegian lake dominated by roach, Braband et al. (1984) found that they fed on both zooplankton in the epilimnion during the night and on the benthos in the littoral zone during the day. Interestingly, Lappalainen et al. (2005) found that they were a significant predator on blue mussels (*Mytelis edulis*) in the brackish waters of the northern Baltic Sea.

Maturation, spawning and fecundity: In the UK, 50% of roach were mature by age three and 100% by age four (Mann 1973). In the warmer waters of Australia, maturation is earlier with females attaining maturity at 2 to 3 years old and males a year earlier (Merrick and Schmida 1984). The males acquire nuptial tubercules on their pectoral fins prior to spawning (Cadwallader and Backhouse 1983). Spawning substrate in Europe is primarily small stones and gravels (Holcik and Hruska 1966; Holcik 1967) and roach tended to use this even if plant material was available. Lacustrine populations that migrate to tributary streams to spawn are no doubt seeking more suitable substrates than found around the lake edge. Spawning occurs over a 10 day period and most eggs are deposited in water less than 20 cm deep (Holcik 1967). Pihu and Kangur (2001) found that roach start to spawn at water temperatures of 8-10°C and that mass spawning occurs at temperatures of 10-13°C. However, spring and autumn spawning populations can occur. In a study on the effects of climate warming (between 1962-1997) on the spawning of fish populations, Noges and Javet (2005) found a decrease in the time of spawning for bream but no change for roach. Instead, the temperature at which roach first spawned increased; suggesting that time of spawning in roach is controlled more by photoperiod than by water temperature.

In Australia, roach spawn in shallow waters over vegetation or on stony substrates between October and November (Weatherley and Lake 1967). The 1-1.5 mm diameter eggs are attached to vegetation or hard objects (e.g., tree roots, stones) and take approximately 4-10 days to hatch depending on the water temperature. The 5-6 mm long newly-hatched larvae are demersal (Froese and Pauly 2007) and attach themselves to the sides of plants before becoming free-swimming (Cadwallader and Backhouse 1983)

Females can produce 5,000-200,000 eggs depending on their size and condition (Cadwallader and Backhouse 1983; Froese and Pauly 2007). Mann (1973) reported a fecundity/length relationship for roach that indicated a 150 mm long female can be expected to produce about 5,000 eggs. At a length of 300 mm she will be producing about 50,000 eggs. Lappalainen et al. (2006) found that female growth was lower, life span longer, and overall reproductive output lower in populations near the latitudinal limit of this species' geographic range, compared with populations near the centre of this range.

Population size and structure: The age structure of European populations varies widely because of large variations in year class strength. For example, Holcik (1967) found that there were eight year classes present in a Czech reservoir, and that in 1964 the cohort of five year old fish dominated the adult population. Mann (1973) also found that large variations in year class strength and growth rate influenced the age structure of roach in the Frome and Stour Rivers in the UK. Between 1963 and 1965, fifteen year classes occurred in the Frome and thirteen in the Stour. The estimated population doubling time for roach ranges from 1.4-4.4 years (Froese and Pauly 2007). Mann (1973) reported a sex ratio close to 1:1 but noted that other studies had found ratios favouring females (1: 2.5). In a Norwegian lake, with a large population of roach, the biomass was estimated at 500 kg ha^{-1} (Braband et al. 1984).

Uses: Roach are a component of the commercial catch in some European lakes and are also caught by 'coarse' fish anglers. They were probably introduced to Australia more for their recreational value (McDowall 1996) than for any potential commercial opportunity.

Impacts overseas: Roach have been introduced to Spain, Portugal, Ireland, Madagascar, Morocco, Kazakhstan, Cyprus, Italy and the Azores Is, and are regarded as a nuisance in areas where they have been introduced and become established (Froese and Pauly 2007).

Griffiths (1997) described the introduction of roach to Ireland by an angler using it as live bait and he indicated that it had spread throughout the country and is now regarded as a nuisance. This appears to be because it quickly dominates the fish fauna where it occurs. The main biotic impact of its introduction in Irish waters has been a decline in rudd (*Scardinius erythrophthalmus*), with hybridisation between rudd and roach likely to be responsible for this (Cragg-Hine 1973; Griffiths 1997). Although roach have increased the abundance of fish-eating birds such as the great crested grebe *Podiceps cristatus* and cormorant *Phalacrocorax carbo* such that their abundance increased (Winfield et al. 1994), they can potentially compete with other waterfowl for small invertebrate prey (Winfield et al. 1992; Giles 1994). Roach have recently been spread to Scotland by anglers (Treasurer 1990) but there are no known reports on impacts at present..

In coastal (low salinity) areas of northern Baltic Sea, roach are thought to be a major predator of the blue mussel which dominates the benthic biomass and hence is a major food base food for many fish and bird species in this region. Lappalainen et al. (2005) estimated that roach predation could account for a third of total mussel production in this environment.

Roach can increase the phosphorous and trace element concentrations required by planktonic algae in lakes (Braband et al. 1984; Braband et al. 1990; Horppila 1998) and so are thought to play a pivotal role in the eutrophication of many small European lakes. Horppila (1998) found that phosphorus excretion by roach in a Finnish lake accounted for 18% of the annual external loading, and that mass removal of roach reduced phosphorus excretion rates by 75%. Hence, they can play a significant role in the eutrophication of lakes and their reduction or removal forms the basis for many lake restoration programmes in Europe.

Introduction and impacts in Australia: Roach were imported from Europe to Tasmania between 1860 and 1870 and then introduced to Victoria (Yarra River drainage), New South Wales, Tasmania and Western Australia (Cadwallader and Backhouse 1983). They were initially restricted to the Yarra River system in Victoria, but were later spread to the southwest of the Great Dividing Range (Cadwallader and Backhouse 1983). The Victorian Department of Primary Industry indicated that roach are now in the Goulburn River system, north of the Dividing Range (Victorian DPI 2007). Roach were recorded once in New South Wales but are unlikely to occur there now. They are not known to occur in the Northern Territory, Queensland or Western Australia. Although angling (coarse fishing) is still the main use of roach in Australia (chapter 6), their spread may have been assisted through use as live bait (Cadwallader and Backhouse 1983).

There has been little study of the effects of roach in Australian waters, but in the web-based publication 'Fisheries Notes' (Victorian DPI 2007), the Victorian Department of Primary Industry indicated that the roach 'is largely a nuisance fish in terms of competing for both space and food with other more desirable introduced and native angling species'. This assertion is presumably based on the observation that roach densities are relatively high in some locations, and the assumption that a high biomass of introduced fish must be removing production from native fish species. Arthington and Blühdorn (1995) noted the observation by P.S. Lake that roach abundance was high in the Port Phillip Bay Rivers and that impacts on the indigenous biota could therefore be expected in these waters. They also recorded the observation by P. Gehrke that the low abundance of roach in New South Wales was unlikely to result in impacts. Lake (1959) indicated that although roach were primarily bottom feeders in Australian waters, they also fed on terrestrial insects at the water surface during summer. Given their omnivorous dietary capacity, there is little doubt that they would feed on similar prey species to those taken by many native fish species. The assumption that competition for food with native species occurs is therefore reasonable but only when high densities of roach coincide with similar sized native fish and when such high density fish populations are limited by food rather than by habitat. At present, there is no evidence that fish populations in Australian rivers are food limited.

4.5 Yellowfin goby (*Acanthogobius flavimanus*)

Description, subspecies and hybridisation: Yellowfin gobies were studied by Miyazaki (1940) and are small, elongate fish with large heads and a cylindrical body form. They possess the typical two dorsal fins of gobies and the fused pelvic fins which form a sucker disc. They are pale brown with dark slashes and spots covering their flanks and are distinguished from other gobies by their large size and yellow ventral and anal fins (ISSG 2007). Other common names include the oriental goby, spotted goby, Japanese river goby and, in Japan, 'mahaze'. Analyses of mitochondrial DNA in populations inhabiting estuaries in California revealed genetic differences between yellowfin gobies in the San Francisco Bay and those in the more northern and southern estuaries. These DNA differences were consistent with separate introductions of this species to the estuaries (Neilson and Wilson 2005). McDowall (1996) provides a key to the Australian gobies with which the yellowfin may be confused.

Indigenous range: The yellowfin goby occurs naturally from the southeastern coast of Russia, around the coast of Japan and Korea, down to northern China (University of California Berkeley, Digital Library Project http://elib.cs.berkeley.edu). Froese and

Pauly (2007) indicated a longitudinal range of 52ºN to 23ºN and a latitudinal one of 116ºE to 143ºE.

Habitats and migration: The yellowfin goby is a demersal species reported to occur on soft-bottomed (mud and sand) habitats in bays and estuaries (Froese and Pauly 2007). However, it also occurs in sea-grass beds (Huh and Kwak 1999), and both Pollard and Hutchings (1990) and Lockett and Gomon (2001) found yellowfin gobies on hard substrates around Sydney Harbour. Their depth range is reported to be 1-14 m (Barnham 1998).

Workman and Merz (2007) found yellowfin goby throughout the tidal portion of the Sacramento-San Joaquin river system. It is apparent that some adult gobies also occur in the lower reaches of rivers where they inhabit brackish waters (ISSG 2007). Upstream penetration of estuaries and rivers is achieved at the larval stage. Larvae disperse rapidly once the yolk sac is absorbed and utilise tidal currents to move upstream and into estuaries. The larvae occur in near-surface waters on a flood tide, then descend to lie close to the bottom on ebb tides. This change in distribution, related to changes in tidal flows, no doubt assists the upstream dispersal of yellowfin goby larvae within the tidal regions of rivers.

Juveniles are epibenthic, often inhabiting burrows, and they occur in shallow marine waters as well as in the lower reaches of rivers, irrigation ditches and canals (http://elib.cs.berkeley.edu). The pelvic fins fuse into a sucking disc which enables juveniles to attach to substrates (http://elib.cs.berkeley.edu) and this presumably allows them to maintain position against the flow.

Tolerances and limiting factors: Little is known about the tolerances of adult yellowfin goby to water temperatures, but warm winter temperatures (>13°C) may limit egg incubation (Dotu and Mito 1955; Bell et al. 1987). Reports of heavy mortalities in reservoirs and lagoons following a reduction in salinity (Nico and Fuller 2007) suggest that, while the yellowfin goby can live in a wide range of salinities, its osmoregulatory ability may be unable to cope with a sudden drop in salinity (from brackish water levels to concentrations less than 5 ppt). Low salinity may therefore prevent this species from inhabiting pure freshwater habitats and so limit its upstream penetration in rivers. It has, however, been reported in freshwater habitats in some rivers; data on actual salinity levels where this species is both present and absent in rivers are required to test this.

Predators, parasites and diseases: Other, larger fish represent one set of the yellowfin goby's natural predators (Froese and Pauly 2007). Yellowfin goby were the most common prey for pied cormorants (*Phalacrocorax varius*) nesting in trees surrounding Lake Borrie, south-west of Melbourne (Ball 1991).

Myxobolus acanthogobii is a myxosporean parasite that infects the brain of the yellowfin goby. It is not known to cause problems for this species but it is synonymous with *Myxobolus buri*, a myxosporean parasite that infects the brain of the yellowtail (*Seriola quinqueradiata*), causing spinal curvature and resulting in significant economic losses for yellowtail aquaculture (Yokoyama et al. 2004).

Age, growth and size: Maximum size is 30 cm and the life span is estimated to be 13 years (Froese and Pauly 2007).

Feeding and diet: The pelagic larvae (9-13 mm SL) of yellowfin goby consume planktonic prey and settle onto the substrate between 13-14 mm SL. These benthic juveniles consume small epiphytic crustacea and polychaetes, with polychaetes predominating in the larger (5-11 cm SL) fish (Kanou et al. 2004; Froese and Pauly 2007). The larger gobies also eat amphipods, mysid shrimps and small fish in marine environments (http://elib.cs.berkeley.edu). Diet in the freshwater-influenced, lower reaches of the Mokelumne River included chironomids, gammarids, isopods, and ephemeropterans (Workman and Merz 2007). Barnham (1998) indicated that the yellowfin goby fed heavily on small fish.

Maturation, spawning and fecundity: The yellowfin goby is oviparous and the spawning season encompasses winter and early spring (Dotu and Mito 1955). Sexual maturity is attained in San Francisco Bay within a year at a size of 270 mm (Baker 1975, cited by Bell et al. 1987). Adults in brackish water environments migrate downstream to spawn in estuaries (ISSG 2007), or on the tidal mudflats abutting coastal areas. Spawning occurs at temperatures ranging from 7.5-13°C (http://elib.cs.berkeley.edu) and eggs are deposited at the base of a Y–shaped nest constructed in a tunnel or burrow (Froese and Pauly 2007; ISSG 2007). Main spawning substrates were sand and mud, but debris such as bamboo segments and ceramic tubes were also utilised. Fecundity was reported to range from 6,000-32,000, with a 156 mm TL female containing 18,000 eggs. The eggs are 5-6 mm long and approximately 1 mm wide and are attached to the roof or wall of the breeding chamber in a single layer. Eggs take 28 days to develop at the optimum temperature of 13ºC (ISSG 2007). The larvae are 5 mm long on hatching, and whilst free-swimming, they were only sampled from waters near the estuary bottom, never in surface waters.

Population size and structure: The population of yellowfin goby expanded rapidly after its introduction to San Francisco Bay (Nico and Fuller 2004), and it also spread rapidly throughout Port Phillip Bay in Australia following its introduction there. However, its abundance in Australia appears to be lower than in San Francisco Bay (Middleton 1982).

Uses: In Japan, fishing for yellowfin goby is a traditional practice that is still carried out today (www.fishingfury.com/a-different-kind-of-fishing/). No fisheries have developed in San Francisco Bay (USA) where this species has recently been introduced and become abundant. Because of its small size, the yellowfin goby is unlikely to be targeted by commercial or recreational fisheries in Australia.

Impacts overseas: Froese and Pauly (2007) reported that although introductions had been reported in the USA, Mexico and Australia, the ecological effects of these was unknown. Nico and Fuller (2007) summarised information relevant to the introduction of this species to the coastal and inland waters of California. They noted the rapid increase in population size of this species once it became established in San Francisco Bay, and the fact that high mortalities had occurred in a reservoir and a lagoon, possibly because of a sharp drop in salinity. Impacts on the native fauna included the partial displacement of a sculpin (*Leptocottus armatus*), and a reduction in the shimofuri goby (*Tridentiger bifasciatus*). Moyle (1976) also expressed concerns over the potential to affect the endangered, tidewater goby (*Eucyclogobius newberryi*). This issue was also raised by the Sacramento Fish and Wildlife Office, who observed that competition from the yellowfin goby would add to the impacts caused to the tidewater goby's habitat by upstream water diversion, dredging, pollution, siltation, and urban development (SFWO 2007). The ISSG (2007) database lists the yellowfin goby on its website and the factsheet for this species reiterates these concerns.

Introduction and impacts in Australia: The yellowfin goby was initially reported from Sydney Harbour in 1971 (Hoese 1973) and it is now well established in Sydney Harbour and Botany Bay (Pollard and Hutchings 1990; Arthington and McKenzie 1997). Pollard and Hutchings (1990) noted its widespread occurrence from the Hawkesbury River, 30 km north of Sydney, down to Port Kembla, 90 km south of Sydney Harbour. A survey of Sydney Harbour in 2002 (AMBS 2002) found that it occurred primarily in the upper estuarine regions of the Parramatta River and Lane Cove River. It now also occurs in the freshwater reaches of the Hunter River as well as the Hawkesbury River (Middleton 1982; Bell et al. 1987; Hutchings 1992). Although yellowfin goby were recorded in Port Kembla, south of Sydney (Pollard and Hutchings 1990), none were found there in 2002 (Pollard and Pethebridge 2002).

Yellowfin goby were first recorded in Port Phillip Bay, Victoria, in 1991. They were mainly in the lower reaches of the Yarra River, below Dights Falls (Hoese and Larsen 1994; Victorian DPI 2007). Specimens were later found in the Maribymong and Werrimbee Rivers, and in Corio Bay (Parry et al. 1995), all within the Port Phillip Bay catchment.

The invasion of the yellowfin goby into Australian waters appears to be more muted than its spread in San Francisco Bay (Brittan et al. 1970; Middleton 1982; Bell et al.

1987). Middleton (1982) attributed this to the relatively warm water temperatures in this region. However, she expressed a concern that, as a carnivore, it may eliminate smaller gobies such as *Arenogobius bifrenatus*, and the juveniles of whiting (*Sillago* sp.); additionally, it may compete with other gobiids as well as bothiids and platycephaliids in the estuarine reaches of the harbour. There is also a concern that it may compete with juvenile flounder (*Pseudorhombus arsius*) and the dusky flathead (*Platycephalus fuscus*) (CSIRO Marine 2007). The high density of this species observed in the lower freshwater reaches of the Hawkesbury River (Pollard and Hutchings 1990) is also cause for concern as the vulnerable juvenile stages of many diadromous species of freshwater fish need to traverse this reach en route to adult habitats further upriver. Despite such concerns, we found no reports or studies on the effects of the yellowfin goby in Australian ecosystems on other marine or freshwater fish.

Warm winter temperatures may well prove a major barrier to the northward spread of this species in Australia (Middleton 1982). Optimal egg incubation occurs at temperatures of 13°C (Dotu and Mito 1955), but the tolerance of eggs to higher temperatures is unknown. If temperatures much over 13°C greatly increase egg mortality, then its spread to more northern waters will be restricted. However, water temperatures will not restrict its potential spread to estuaries and harbours in South Australia and Tasmania.

4.6 Streaked goby (*Acentrogobius pflaumii*)

Description, subspecies and hybridisation: This species, formerly known as *Ctenogobius pflaumii* and *Rhinogobius pflaumi* (Matsumiya et al. 1980), was initially identified as *Amoya pflaumii* in the Port of Geelong (Currie et al. 1998). Francis et al. (2003) indicated that the taxonomic status of species in the genus *Acentrogobius*, and its related genera, is uncertain. Lockett and Gomon (2001) noted that their specimens from the Port of Melbourne had 10 dorsal fin rays as did the syntypes of this species from Nagaskai. However, other Asian populations of this species are reported to have nine second dorsal fin rays.

The streaked goby, also known as the striped and Asian goby, has an elongate, tubular body form with a large head and the characteristic two dorsal fins. Coloration is cryptic, and in some fish, small, dark-coloured patches on the flanks are horizontally aligned to form a stripe. The ventral and lower flank regions are a uniform pale white or sandy colour. McDowall (1996) provides a key to the identification of other endemic Australian gobies with which the streaked goby may be confused.

Indigenous range: The streaked goby has been reported from the north-west Pacific, down through Korea and Japan to Taiwan and the Philippines (Francis et al. 2003). Froese and Pauly (2007) also report its occurrence in Russian coastal waters; however, given the uncertainty over its taxonomy (Francis et al. 2003), the northernmost records could involve a different species.

Habitats and migration: In its native range, the streaked goby occurs on flat, sandy or muddy substrates close to *Zostera* sp. (sea-grass) beds, and it can also be found within the *Zostera* sp. beds, suggesting movement between these two habitats (Matsumiya et al. 1980). Horinouchi and Sano (2001) found that new recruits (22 mm SL) appeared first in *Zostera* sp. beds in autumn and were absent on bare substrates. They concluded that their occurrence in the *Zostera* sp. beds was mainly because of the availability of suitable prey species and not because of protection afforded against predators. In contrast, adults predominate in the open, more exposed, soft substrates beyond the *Zostera* sp. beds. In Australia and New Zealand**,** the streaked goby occupies similar habitats (Francis et al. 2003) to those outlined above for this species in Asia. In Port Phillip Bay, adult streaked goby were rare in the shallows and more common in waters over 5 m deep (Hamer et al. 1998). This has also been observed for streaked goby in western Australia (Maddern & Morrison, In press). Co-habitation of burrows with alpheid shrimps has been observed only in Western Australia (loc cit.)

Tolerances and limiting factors: The geographic distribution of the streaked goby indicates a relatively wide temperature tolerance (temperate to subtropical). It occurs in brackish waters in the lower reaches of rivers, including the lower reaches of the Yarra River (see Francis et al. 2003), so clearly has some tolerance to lower than full-strength salinities.

Predators, parasites and diseases: Froese and Pauly (2007) indicated that the main predators of the streaked goby in Korea and Taiwan were piscivores, including barracuda and sole. No information was available on parasites.

Age, growth and size: The planktonic larval stage is reported to be 30 days (Lockett and Gomon 1999) and new recruits settle on the substrate in *Zostera* sp. beds in autumn at a mean size of 22 mm **(**Horinouchi and Sano 2001). Growth is rapid and they nearly double their length over the next 2-3 months. Fowler (1960) and Masuda et al. (1975) report a maximum size of 80 mm TL, but the maximum size reported in Froese and Pauly (2007) is 120 mm. The largest fish found in both New Zealand and Port Phillip Bay in Australia were 70 mm (Francis et al. 2003), suggesting slower growth in these somewhat colder waters. Based on the maximum size of 120 mm, Froese and Pauly (2007) estimated a life span of 6 years.

Feeding and diet: Larvae (5-8 mm) are planktonic and feed on copepods (Kanou et al. 2004). Prey species for benthic juveniles (20-50 mm long) in *Zostera* beds included mainly harpacticoid copepods and gammarid amphipods (Matsumiya et al. 1980; Horinouchi and Sano 2001). Adults feed more on benthic invertebrates especially amphipods (Kanou et al. 2004).

Maturation, spawning and fecundity: In Japan, streaked goby mature during their first year of life and spawning occurs over summer months (Mori 1995 in Horinouchi and Sano 2001). Larvae are 5-8 mm long and planktonic (Kanou et al. 2004), and the larval stage is about 30 days (Lockett and Gomon 1999). In Korea, spawning occurred in early summer (i.e., May-June), when fish were 4 cm long (Baeck et al. 2004). Fecundity was 3,600-9,700 eggs per female depending on size (Baeck et al. 2004).

Population size and structure: The densities of juveniles in *Zostera* beds averaged 1-2 fish m^2 (Horinouchi and Sano 2001). Adult streaked goby are now one of the most abundant fishes in the shallow near-shore habitats of Port Phillip Bay (Lockett and Gomon 2001). Baeck et al. (2004) reported that the proportion of females increased with fish size: 60% of all fish were female, including 100% of fish over 5.5 cm long.

Impacts overseas: None are reported in Froese and Pauly (2007), the US Geological Service database on invasive species, nor in the Invasive Species Specialist Group (ISSG 2007) database.

Introduction and impacts in Australia: The streaked goby was first recorded near the Victoria Docks in Port Phillip Bay, Melbourne in 1996 (Knuckey et al. 1997; Lockett and Gomon 1999). Subsequent sampling indicated that it was present throughout most of the Bay (Currie et al. 1998; Pollard and Pethebridge 2002) and it was the eighth most abundant fish in beam trawl catches (Hamer et al. 1998). It was subsequently reported inhabiting Botany Bay and Sydney Harbour (Francis et al. 2003).

Although the streaked goby became abundant and widespread in Port Phillip Bay within a decade of its introduction (Lockett and Gomon 1999; 2001), its impact on marine life and the environment is unknown (Francis et al. 2003).

Maddern & Morrison (In press) found streaked goby in micro-habitats also frequented by *Arenigobius bifrenatus* and indicated that, becausee of such spatial overlap, there may be potential for competition to occur.

4.7 Combinations of alien species

One of the difficulties in determining the ecological impact of an introduced fish species on the native fauna is that the introduction is rarely the sole change in the ecosystem that could account for a decline in indigenous biodiversity. Arthington et al. (1993) found that habitat degradation often characterised sites where introduced species thrived and, in waters around Brisbane, gambusia often occurred together with the green swordtail, *Xiphophorus helleri*, such that the impact of gambusia alone could not be determined. Similarly, trout and redfin perch often co-occur in the lower reaches of rivers, and the streaked and yellowfin gobies both occur in the waters of Sydney and Melbourne Harbours.

Although co-occurrence of introduced species creates difficulties in impact assessment, it also raises the prospect of synergistic effects whereby the effects of one introduced species are compounded and increased by the presence of another. The green swordtail has affected native fish species in the USA (Courtenay et al. 1988) and Morgan and Gill (2001) indicated that it had displaced other fish species in the Irwin River. It can dominate *G. holbrooki* (Milton & Arthington 1983; Arthington et al. 1986) so it is clear that it could well exacerbate the impact of gambusia on native fish in Australian waters. Warburton and Madden (2003) found that attacks by gambusia on native species in tanks increased when *X. helleri* was present, and were intensified when food was added. Such results clearly indicate the potential for synergistic effects of introduced species combinations on native fish in the wild.

Similarly, the disappearance of native fish species from the Big Brook Dam was attributed to the combined effects of redfin perch, gambusia and trout (Morgan et al. 2002). Whereas predation by redfin perch can be expected to be the main factor, the presence of rainbow trout and gambusia may well have reduced the scope for native species to find refugia, augmenting the mortality directly attributable to redfin perch. In addition, their presence in a range of habitats will have increased competition for food.

Other opportunities for synergistic impacts are suggested by the observation that large redfin perch preyed heavily on marron (Morgan et al. 2002), which are cited in other studies (loc. cit.) as major predators of gambusia. Hence, the presence of redfin perch may well increase the impact of gambusia on native species by enhancing gambusia survival.

Roach and tench are neither piscivorous nor aggressive consequently their impacts on native fish in Australia will not be as direct and significant as those of gambusia and redfin perch. Impacts may occur through competition for food. However, where non-piscivorous fish species from different feeding guilds (e.g., planktivores, herbivores, detritovores) are abundant there is scope for impacts on water quality in lakes and reservoirs, and hence habitat changes. Gerhke and Harris (1994) reviewed the role of fish in the creation of blue-green algal blooms in Australian lakes, and noted that planktivorous species including redfin perch could reduce the controlling effect of *Daphnia* on phytoplankton densities in lakes; similarly, carp, roach and gambusia were all capable of influencing the nutrient dynamics in lakes and ponds such that blue-green blooms could be exacerbated. Rowe (2007) found that many small New Zealand lakes containing introduced fish (e.g., redfin perch, tench, carp, goldfish, catfish) usually contained more than one species of introduced fish, and that water clarity was lower in lakes with introduced fish species than in lakes without them. However, as water clarity was lower in lakes with three or more introduced fish species than in lakes with only one or two species, it was apparent that the presence of fish from several different feeding guilds accelerated the eutrophication processes.

It is becoming increasingly apparent that the presence of more than one introduced fish species in an aquatic environment can result in synergistic effects on water quality, as well as on the survival and habitat of native fish species. That such synergistic effects are now being identified more frequently underscores the urgent need to restrict the spread of introduced fish to waters already affected by other alien species.

5. Genetic impacts

5.1 Introduction

This section discusses the genetic threats posed to native fish fauna by the introduction of tench (*Tinca tinca*), roach (*Rutilus rutilus*), redfin perch (*Perca fluviatilis*), gambusia (*Gambusia holbrooki*), yellow finned goby (*Acanthogobius flavimanus*) and streaked goby (*Acentrogobius pflaumii*). 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). The main genetic threats to native fish fauna are likely to be 1) hybridisation and introgression, and 2) problems associated with small populations due to deleterious ecological interactions and disease. Additionally, 3) hybridisation between alien fish taxa may have some serious consequences, for both species biology and management. Please note that it was not possible to use fish to illustrate all points in the following discussion; however, case studies involving fish have been used wherever possible.

5.2 Hybridisation and introgression

Historically, hybridisation 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 and Childs 1992).

5.3 Isolating mechanisms

To better understand the threat posed by the hybridisation of endemic and introduced fish fauna, we need to understand both the mechanisms that increase 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.

Pre-mating isolation: 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 i) phenotypic, ii) temporal, iii) ecological, and iv) ethological preferences.

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, based on appearance, species may be thought to share a similar evolutionary lineage, but have in fact 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 species may be distinguished. 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^{o}C to 23^{o}C (Harris and Rowland 1996). The congeneric trout cod, however, spawns earlier in the season at a slightly lower temperature (Cadwallader and Lawrence 1990). These preferences are likely to keep both congenerics separate during the spawning period. However, both species have been known to hybridise in the wild (Douglas et al. 1995), and 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, spatial differences in spawning habitat are 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 and 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 and Rowland 1996). These life history differences are very effective at isolating each 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. To take two avian examples, 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 decorates 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 and 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 (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 was available for one sympatric species or no closely related taxa coexist. For example, pre-spawning courtship has been observed for eastern freshwater cod (*Muccullochella ikei*), but there are no data for Marry 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 of these last three species has still not been witnessed in the wild (S. Rowland pers. comm.).

Post-mating isolation: Many groups of aquatic taxa, such as fish, sponges, corals, bivalves, ascidians and echinoderms, have no courtship behaviour: being external spawners, they release their gametes en masse. For some groups (corals are a good example), group spawning takes place under certain environmental conditions, and many species have synchronised gametic release. As a result of this mass 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 diverge through time, and isolation. Some species have developed self-compatibility mechanisms that can actively reject gametes if they are incompatible (Kao and 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 and Johnstone 1994). Such differences between taxa are likely to pose a significant barrier to reproduction between them. Additionally, as these isolated species/populations move through evolutionary time and space, these reproductive incompatibilities are likely to become more pronounced. Post-mating isolation, observed as sperm/egg incompatibilities, has been reported in aquatic invertebrates, such as sea urchins (Palumbi and 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 - similar enough to have once been classified as different morphotypes of the same species. Despite these similarities (molecular evidence suggests that they most likely shared a direct common ancestor), strong incompatibilities during sperm-egg attachment prohibited fertilization. In such cases, species boundaries are not crossed, reinforcing species boundaries.

Post-zygotic isolation: 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 if F_1 progeny are fertile, backcrosses with parental species may be halted by incompatibilities between the hybrid and parent (Rhymer and Simberloff, 1996). In both of 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 native (to North America) 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. This 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 is the case with Apache trout (*Oncorhynchus gilae apache*). In this instance, genes from translocated rainbow trout (*Oncorhynchus mykiss*) have introgressed into Apache trout genomes, but the reverse has not occurred (Dowling and 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).

5.4 Likelihood of hybridisation between native and introduced fish fauna

The biological species definition that delineates species as being reproductively isolated from all other species (Mayr 1963) is not perfect; indeed, many species, especially plants (Gillet 1972; Levin et al. 1996) and fish (Hubbs 1955; Avise and Saunders 1984; Rubidge and Taylor 2005) hybridise continually. Moreover, hybridisation is likely to be an important mechanism in the evolutionary process. The major determinant for the likelihood of hybridisation and introgression between species – more telling than any other factor – will be their evolutionary relatedness. For it is incompatibilities at the chromosomal and genetic level that will prevent the production of offspring. Fortuitously, the Australian fish fauna is highly endemic and does not contain major groups common to most other large land masses. For example, whereas the native Australia freshwater fish fauna includes species from 38 families, these do not include the Poeciliidae, Percidae or Cyprinidae to which most of the alien species belong. However, Australia does contain native fish in the Gobiidae family, which also includes the introduced yellowfin and streaked goby.

The likelihood of the introduced taxa hybridising with the native species will be a function of relatedness, habitat and behavioural overlap, and gametic compatibility. As we has have seen, the closer the evolutionary relationship between two species, the higher the chance of hybridisation and introgression. Despite both introduced gobies belonging to the same taxonomic family, they are not members of any endemic genus. There is no current broad-scale phylogenetic tree that includes both the introduced and endemic taxa, which makes any direct estimation of evolutionary relatedness, beyond the overarching family level, a difficult task. To assist in clarifying this relation, future research into the phylogenetic relationships between these taxa should be encouraged.

Both introduced gobies have similar habitat preferences to native goby species, and the extent to which there is overlap will depend on the extent to which either species spreads around the coast. Given that these species are likely to have been transported in ballast water or within oyster shipments, it seems plausible that such practises will allow the future spread of these fish within Australia. Researchers have suggested additional vectors for movement, including the movement of eggs attached to encrusting organisms on ships' hulls (Nico and Fuller 2004), and in recreational craft, as some locations where these species have been found are not commercial shipping ports (Lockett and Gomon 2001). Therefore, given their habitat preference and likely spread, the chances of these introduced taxa coming into contact with native members of the same family could be significant.

The taxonomy, life history and reproductive biology for many endemic gobies remains unclear. However, gobies are known to create nests to deposit eggs. These nests may be among shell fragments, and may consist of burrows and/or tunnels. The introduced yellowfin goby are also known to create y-shaped nests (15-35cm deep) in lower estuaries by burrowing into the sediment (Barnham 1998; Global Species Database 2005); the nesting behaviour for the streaked goby is not well understood. The similarities in breeding behaviour within the goby family, including external fertilisation, presents at least some risk of hybridisation. However, the yellowfin goby is known to be aggressive to other species (Barnham 1998), which may reduce reproductive interactions, especially if the species aggressively maintains a territory.

Thus, the genetic threats of hybridisation, introgression, and the dilution of species boundaries must be considered negligible for gambusia, redfin perch, roach, and tench (Table 5.1). However, given the uncertainty involved, a rating of medium for the yellowfin and streaked gobies would be applicable.

Table 5.1: Rating of the risk of genetic impacts for the species reviewed.

Species	Threat
Gambusia (*Gambusia holbrooki*)	Low
Redfin perch (*Perca fluviatilis*)	Low
Tench (*Tinca tinca*)	Low
Roach (*Rutilus*)	Low
Yellow finned goby (*Acanthogobius flavimanus*)	Medium
Streaked goby (*Acentrogobius pflaumii*)	Medium

5.5 Genetic effects of demographic contractions as a result of interactions with introduced fish fauna

Interactions between native and alien species are likely to be negative in many ways (Costedoat et al. 2005; Gurevitch and Padilla 2004). These negative interactions in some situations have the potential to reduce or fragment native populations (Wayne et al. 1992). For example, gambusia may 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.

5.6 Negative effects of small population size

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 and 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 (Soulé 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 and Ballou 1983; Gall 1987) and include decreases in individual and Darwinian fitness (Wright, 1977), and increases in deformed offspring (Kincaid 1976a; 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 and 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 and Soulé 1981).

5.7 Hybridisation between introduced fish fauna

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.

Hybridisation and evolution: There can be little doubt that hybridisation contributes to the evolutionary process. From the Neodarwinian 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 affects of hybridisation are the production of infertile offspring due to post-zygoting isolating mechanisms, and reduced recruitment, resulting from gametic incompatibilities or the breakdown of stable embryological pathways (Rhymer and 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), fish (Hubbs 1955; Avise and Saunders 1984; Rowland 1984; Campton 1987; Baker et al. 2002; Rubidge and 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 and 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. Most small populations lose genetic variation through genetic drift faster than it can be maintained through mutation, and 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 the population/species through evolutionary time (Frankel and Soulé 1981). A loss in adaptive potential increases the risk of extinction (Soulé 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 fish are likely to have been through at least one significant founder event and 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 and 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 and Mulley 1978). Despite the long-term persistence of both these populations (introduced by 1908, although their introduction may have been as early as the 1860s), it was the liberation of the Boolara strain in the 1970s that resulted in the large-scale spread of the species throughout Australia (Morison and 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, although cane toads and gambusia are two notable examples of founder populations being aggressive adaptors. It must be noted that the impact of bottlenecks is a function of the severity and length of the contraction. 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 are 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 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 and the breakdown of co adapted gene complexes. Forms of outbreeding depression can be seen in anandromous salmonid fishes (Gilk et al. 2004). Hybridisation has had a detrimental affect on spawning timing, ability to find suitable spawning habitat, orientation of newly emerged fry, and overall reproductive fitness (Rhymer and Simberloff, 1996). Granath et al. (2004) found higher survival rates in control lines of Alaskan salmon (*Oncorhynchus kisutch*) than in hybrids formed by crossing geographically separate populations of the species. Such changes can erode fitness and weaken a population; in some cases this can be catastrophic if the selective pressure on a 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).

5.8 Likelihood of hybridisation between introduced fish fauna

The six species under discussion here represent three distinct families, two of which are non-indigenous to Australia. Hybridisation and introgression within each family is possible where multiple members of the same family occupy the same environment and share similar breeding characteristics. For the family Cyprinidae this has already occurred. The consequences can be quite significant, but due to a paucity of research in the area, these consequences will all too likely go undetermined.

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 (*Barbus (Puntius) conchonius*), roach

(*Rutilus rutilus*) and tench (*Tinca tinca*). Crussian carp (*Carassius carassius*) have also been recently confirmed as occurring in Australia (Davies et al. 2008; Raadik 2007). Hybridisation has been reported between goldfish (*Carassius auratus*) and European carp (*Cyprinus carpio*) throughout Victoria, including drainages of the Murray (Hume et al. 1983). Hybrids between Yanco strain carp and goldfish have been detected in the Murrumbidgee Irrigation Area in New South Wales (Shearer and Mulley 1983), as have intraspecific hybrids of Yanco and Boolara strain carp (Mulley and 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 cyprinids commonly hybridise (Costedoat et al. 2005). The evidence that this group can and does hybridise suggests that we may well see further examples as more 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: the sailfin molly (*Poecilia latipinna*), Guppy (*Poecilia reticulata*), green swordtail (*Xiphophorus hellerii*), platy (*Xiphororus maculatus*), one-spot livebearer (*Phalloceros caudimaculatus*), and the mosquitofish (*Gambusia holbrooki*). Poeciliids are known to hybridise in the wild (Hubbs 1955; Scribner 1993; Rosenthal et al. 2003) and in captivity (Scribner and Avise 1994; Lima 1998; Scribner et al. 1999); indeed, the Amazon molly (*Poecilia formosa*) is a recognised hybrid species (Hubbs 1955; Schartl et al. 1995; Lamatsch et al. 2002; Dries 2003; Tiedemann 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.

Percidae: Redfin perch (*Perca fluviatilis*) is the only member of the family Percidae in Australia. However, the species is widely distributed throughout the world (Lever 1996) and therefore would have many subpopulations that have been separated long enough to be genetically distinct from the Australian form. However, the importation of this species is unlikely in the future and therefore presents little threat.

Gobiidae: Both the yellowfin goby and the streaked goby are estuarine dwelling species, and as such there is a significant chance of them occupying the same waters. Both species share a common morphology and are also most likely to share quite similar breeding and life-history traits. Therefore, the chance of reproductive congruence is possible. There is presently insufficient data on the demography and colonisation of either species to predict the threat of hybridisation.

5.9 Summary

Hybridisation, introgression, and the breakdown of species boundaries pose significant threats to biodiversity throughout the world. The old paradigm of 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 appears to be an intrinsic part on the evolutionary process.

The threats of hybridisation, introgression and the breakdown of species boundaries posed by roach, tench, redfin and gambusia are likely to be negligible. This argument is derived form the high level of endemism in Australia's fish fauna and the absence of major families such as Cyprinidae, Percidae, and Poeciliidae. As has been described, the differences between these introduced and native groups are very likely to be sufficient to prevent any form of species crossing. However, the threats posed by the yellowfin and streaked goby are more significant, as the family Gobiidae is present and widespread in Australia. The introduced gobies are likely to share common habitat and reproductive preferences. Therefore, the risks associated with these species are likely to be moderate.

The genetic threats posed by these species are likely to stem from 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, individual and population health. The deleterious consequences of small population size are likely to be increases in inbreeding, the loss of fitness associated with inbreeding depression, and the loss of allelic diversity and heterozygosity. The species/populations most likely to suffer genetically will be those that are reduced to the smallest population size.

Hybridisation within alien taxa has already happened to some degree (the hybrid between carp and goldfish, for example), and has the potential to happen in the future. Hybridisation within alien fish fauna carries the threat of producing hybrids with greater fitness and increased adaptability to expand into new ecological niches. 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 the reproduction, systematics, population genetics, and impacts of introduced taxa in Australia suggests that research priorities need to be focused on this issue if we are to move forward. It is likely that such information will prove essential in the control of these taxa in Australia.

6. Social impacts

6.1 Introduction

There are few recorded studies on the social impacts of pest species. It is known that invasive species in general can have impacts on human health, safety, recreational activity, social infrastructure, job opportunities, quality of life and cultural heritage (Agtrans Research 2005). However, the extent of these effects is unclear and cannot easily be extrapolated to fish because hydrological connectivity places additional limits on ‘invasion’ compared with terrestrial species.

This chapter has been prepared to provide an overview of the social impacts of a number of alien fish species in Australia (roach, tench, redfin perch, eastern gambusia, streaked goby and yellowfin goby). The aim is to provide a baseline snapshot in terms of people’s attitudes to each species across Victoria, New South Wales and South Australia, in order to identify social impacts that need to be considered alongside economic or ecological impacts. The chapter will also inform the development of a social impact assessment tool, which will provide a framework for ongoing consideration of the social impacts of alien fish species.

6.2 Methods

The study involved a series of key tasks, each of which contributed to the Social Impact Assessment (SIA). The tasks included:

Literature Review

Key documents were selected that demonstrated methodologies or previous research into the social impacts associated with the introduction of alien species. Each document has been reviewed for its relevance to this project, and its key messages are set out.

Development of an SIA framework

The SIA framework was developed from our knowledge of other SIAs and key findings from the literature review. This framework provided the basis for discussion during consultation. Based on international social impact indicators, the following themes shaped the development of the SIA.

Theme 1: Way of life

- Impacts on recreational opportunities
- Impacts on local employment/industries
- Impacts on amenity values
- Impacts on property values

Theme 2: Health and wellbeing

- Impacts on health and wellbeing

Theme 3: Culture and community

- Impacts on indigenous cultural heritage and beliefs
- Impacts on community values

Theme 4: Environment

- Impacts on connectivity of waterways/water quality/chemistry
- Impacts on native habitat and species diversity

Theme 5: Fears and aspirations

- Impacts on species populations and ecosystems/biodiversity

Stakeholder consultation

A series of stakeholder workshops were conducted to better understand the attitudes of representatives from ecological, conservation and fishing backgrounds. It was envisaged that perceptions of each species may differ greatly between the stakeholder groups. We therefore felt that holding workshops with each group individually would allow a more informed understanding of these perceptions.

Key stakeholders were selected based on their scientific, environmental or fishing backgrounds in Victoria, New South Wales and South Australia. Where individuals were unable to attend a consultation workshop, telephone interviews were conducted. The aim of the consultation process was to identify data sources that could be used to measure the likely social impacts of alien fish species now and in the future.

As this study provides a national overview of pest fish species, an understanding of regional differences in perceptions of the selected fish species is fundamental to developing a rigorous SIA. Ultimately, the aim is to adapt the final model to all states and in Australia.

Representatives from the following organisations and government departments were consulted as part of this SIA process:

- Department of Primary Industries (Fisheries), Victoria
- Department of Primary Industries (Fisheries), NSW
- South Australian Research and Development Institute
- VR Fish, Victorian Peak Fishing Body
- Native Fish Australia
- NSW Council of Freshwater Anglers
- Sydney Coarse Anglers
- South Australian Freshwater Anglers Association
- Murray Darling Basin Commission

In addition, the project scope included consulting with representatives from environmental/conservation groups to provide a balanced perspective of social impacts. Attempts were made to consult with the following environmental groups via a number of avenues. Although these groups were interested in the project, they did not have resources available to send representatives to the workshops. Therefore this stakeholder group was not represented throughout the consultation. It is recommended that future assessments continue to seek representation from these groups.

These included:

- World Wildlife Fund
- Australian Conservation Foundation
- Wilderness Society
- Greenpeace
- NSW Nature Conservation Council

Key questions asked of the workshop participants included:

- What is your perception of the value of coarse fish (specifically perch, tench and roach) as a recreational fishery?
- What is your perception of the effects of these coarse fish on Australian native fish?
- What is your perception of the effectiveness of gambusia for mosquito control?
- What is your perception of other species for mosquito control compared with gambusia?
- What is your perception of the effects of gobies (streaked and yellowfin) on Australian native fish?
- What is your perception of the social or economic costs/benefits of gobies in the Australian environment?

Participants were asked for information on available data sources or indicators that could be used to monitor the perceived impacts to provide an ongoing measure for the future. The workshop process was a means for discussing the likely social impacts, whether real or perceived, of the selected species' presence in Australian inland waterways. An overview of the project was presented to participants to explain the process of assessing likely social impacts and how the assessment will inform the broader study output.

6.3 Results of the literature review

Calculating the social costs of introduced species and their impact on native fish populations in Australia is difficult and complex. Economic impacts can be calculated through analysing the costs of control and management, and the monetary values of recreational fishing and associated industries. However, estimating social impacts is more problematic and costly, mostly due to the difficulty in finding good quality data.

Some quantitative and qualitative protocols have been developed to monitor the social impacts of deliberately introduced fish species, as reported in various studies reviewed below. The examples discuss the application of various combinations of hazard identification and assessment protocols, adapted to suit the species being investigated. Key findings from these case studies were applied to the development of the SIA framework for this study.

National recreational and indigenous fishing survey: The National Recreational and Indigenous Fishing Survey (NSW Fisheries 2002), undertaken during 2000-2001, provides a comprehensive collection of statistics related to fish catch, fishing effort, and species composition, as well as to the demographic profile of fishers, and their expenditure and attitudes towards fisheries management issues. This data source provides an insight into the value of some of the fish in this study, particularly redfin perch, and provides a baseline database for comparison with future surveys.

Although this survey dates back to 2000, it indicated that a large proportion of Australia's resident population are involved in recreational fishing. It identifies that an estimated 3.36 million people aged over 5 years fished at least once, which is a national participation rate of 19.5%. Fishing participation was highest in NSW (999,000) followed by QLD (785,000) and Victoria (550,000). In terms of economic flow-on effects, the same survey identified that approximately 445,000 Australian residents aged 5 years and over held a licence for recreational fishing activity. Licence ownership in each state was dictated by the current government policy in relation to licensing of recreational fishing. Depending on the dollar value of a licence, this represents a large economic value to fisheries departments. The survey went into detail on the proportion of fishers that preferred freshwater to saltwater and found that 20% of recreational fishing effort occurred in freshwater. In Victoria, there was a greater reliance on freshwater resources compared with the national average, and the ACT, being landlocked, obviously relied predominantly on freshwater fishing.

Of the species reviewed in this study, redfin perch were identified as a popular recreational species. According to the survey, redfin (and carp) have "successfully adapted to their new environment and now represent substantial components of the recreational harvest". In particular, the survey identified that redfin perch were the most abundant freshwater species caught in Victoria. It was also the third most abundant species harvested in Victoria and the ACT.

Expenditure on fishing in Australia during the survey period equated to $1.86 billion. This included accommodation, camping gear, bait/berley, boat/trailer, clothing, dive gear, fees and licences, fishing gear and travel.

Global invasive species programme (GISP)[1]: The Global Invasive Species Programme (GISP) is a program set up to provide insights into the social risk assessment of invasive species. GISP was undertaken to provide an international perspective on the impacts of invasive alien species (IAS) on inland water systems. This assessment provided an overview of socio-economic case studies, including available data and knowledge gaps.

According to the study, invasive species impact on local economies, including numerous industries, such as fisheries, tourism and water production (Ciruna et al. 2004). This assessment highlights the challenges associated with measuring socio-economic impacts, as the same invasive species that cause damage to ecosystems may concurrently produce economic benefits. A balanced approach is therefore required to ensure the appropriate cost/benefit outcome is achieved to maintain ecological and economic sustainability. These choices are not always straightforward and therefore, as the authors of GISP point out, “it is important to understand the relationship between economic choices and ecosystem health so that economic incentives can be used to mitigate the impacts of IAS, and ensure that both ecosystems and economies are safeguarded.” In terms of social impacts, the assessment identified that protecting ecosystems not only protects biodiversity, but also contributes to the protection of human health, production standards, and access to overseas markets.

Socio-economic impacts can be broadly categorised into market (changes in prices) and non-market (changes in ecosystem services) impacts. Market impacts are caused by production losses due to decrease in fisheries and aquaculture as well as decreases in the availability and accessibility of water industries, decreases in the navigability of lakes and rivers, and declines in property values (Ciruna et al. 2004). Non-market impacts include a number of risks to human, natural and social capital,[2] including impacts on community interactions, severe health impacts, premature death, and loss of ecosystem services due to decline of natural capital (Ciruna et al. 2004).

Internationally, many people depend on inland freshwater ecosystems for their survival; any impacts on these systems have potential to cause adverse social consequences. Some ecosystems are more sensitive to impacts than others, or are more prone to invasion as a result of human disturbances, leading to more costs. The GISP approach highlighted that invasive species will affect different sectors of society in different ways, depending on “where they live, their source of livelihood, and the range of control and eradication strategies available to them” (Ciruna et al. 2004). For example, people on lower incomes may be affected more severely by invasive species than those at a higher income level. This is likely to impact on the types of management and control methods being sought by the community, because

[1] Developed by Ciruna et al. (2004)

[2] Social capital is defined where the relationship between people is thought to be integral to sustain trust in societies. Ciruna et al. *(* 2004).

subsistence level producers may value control more than those on higher incomes not affected by decline in native fish production.

GISP noted that invasive species are often not identified until they have become a problem, which causes a lag effect, posing issues for intergenerational equity. Although there may be benefits for the current generation, there may also be large costs on future generations. A number of case studies were identified, including an invasive riparian weed *Tamarix* sp. that had been introduced as an ornamental plant to South West United States, Australia and Mexico from Eurasia. Although the plant was introduced over one hundred years ago, only recently has its damage to native plant species been discovered. This species was found to have economic impacts on municipal, agricultural, hydroelectric power generation, and river recreation sectors, demonstrating market and non-market impacts. Economic losses due to the invasion of *Tamarix* were significant, which ultimately led to the decision to control the species, rather than find an alternative source of water.

Risk assessment for the wet tropical bioregion[3]: Within Australia, Webb (2006) analysed risk assessment models for non-native freshwater fishes in the Wet Tropic Bioregion. In this assessment, hazard identification was undertaken by determining the life history characteristics of existing and previous invading species, compared with species assumed to be non-invasive. The hazard assessment phase focussed on ecological impacts but also provided for broader, environmental, social and economic values of a specific region in relation to specific target species and the potential threats they pose. Species were ranked based on the nature or severity of the threat. Webb noted that this approach relies on expert opinion and tends to be more subjective given the difficulties in quantifying impacts of target species. In order to assess uncertainty, a series of questions were posed, designed to establish the risk of introduction (deliberate/unintentional), the risk of establishment (climate matching with donor region), and the risk of other significant impacts (economic, environmental, social, dispersal and spread). Each question was given a score based on the total of all questions answered. The percentage of questions that were unanswered was used as a measure of uncertainty regarding the assessment of the species.

Webb recognised the lack of detailed socio-economic impact assessments for other non-native fish species in Australia. The likely socio-economic impacts identified included downturn in local economies (through loss of ecotourism), recreational fishing opportunities, amenity values and employment opportunities. Also mentioned were perceptions of fisheries management and associated frustration with control measures. This study links in with the issues identified by Ciruna et al. (2004) in their study on the need for monitoring of introduced species in order to prevent the species becoming a pest, and to reduce the lag effect that causes intergenerational inequity.

[3] Developed by Webb (2006)

Another important social impact that has had little consideration is the impact of non-native fish species on indigenous communities, as maintaining connections with traditional country and culture are fundamental to indigenous people's identity and well being. Fisheries resources are an important part of indigenous communities' food, culture, spirituality, trade, health and education.

Impact assessment of common carp in Australia: McLeod (2004) used a triple bottom line approach to the impacts of pest vertebrates in Australia. According to this report, the social impact is the hardest to define and quantify. In analysing the social impacts of pest vertebrates, McLeod (2004) considered the effect such species would have on employment prospects within rural and regional Australia, on traffic accidents, and on indigenous Australia. Some positive impacts were identified; however these did not outweigh the adverse impacts.

This approach was used to measure the economic, environmental and social impacts of carp (*Cyprinus carpio)* in Australia. It showed that the impact carp had on the aquatic environment led to detrimental effects on the value of wetlands; these costs were added to management and research costs, to measure the overall economic impact of carp as an invasive species. Costs can be further attributed to the loss of fishing opportunities for recreation where carp is present, whereby fishing is restricted as a management solution. Limiting recreational fishing opportunities affects the wellbeing of fishermen, and reduces the amount of money being spent on support industries. This is apparent in Lake Crescent, Tasmania, where fishing was banned to protect native fish populations as a result of the presence of carp. Carp has also been found to impact on tourism and local commerce through the decline in recreational fishing values, which was identified in a study in the Gippsland Lakes (McLeod 2004).

Conversely, positive social impacts of carp have also been measured, based on the employment opportunities carp provides in rural and regional Australia for commercial harvesting. Carp can be used for fertiliser and crayfish bait. The value of the carp fishing industry was calculated at $1.7 million in 2002 (McLeod 2004). Comments from a stakeholder during the consultation phase of this study, however, identified that this industry is declining. In spite of this, it is still important to consider both positive and negative impacts of alien species.

Murray-Darling Basin report: A study by Wilson (2005) analysed the impacts of invasive alien fishes on wetland ecosystems in the Murray Darling Basin (MDB). Wilson noted previous studies that ranked alien fishes based on stakeholder perceptions of their threat to aquatic habitats. In order, from most threatening to least threatening, these were: carp, gambusia, goldfish, redfin perch and weatherloach. Tench and roach were also listed as posing minor threats due to their localised or minor wetland abundances. According to Wilson, there is relatively little data available on the impacts of alien fishes in the MDB. With limited historical information, this makes measuring future impacts difficult, as there is no reference point to measure change over time.

Wilson (2005) investigated public perceptions and awareness of alien fishes by determining whether each species had a positive, negative or benign impact and then sought feedback on the overall awareness of impacts. The results revealed that of each alien species, carp was the only fish widely perceived to have had negative impacts. Other species such as gambusia were perceived to have a benign impact, or to represent an unknown threat, whereas redfin perch was perceived positively due to its popularity as an angling target, with only slight awareness of its negative impact.

Review of progress on invasive species: This review (Agtrans Research 2005), further recognises the lack of studies that have identified, in specific or qualitative terms, the health, safety and quality of life choices or impacts of invasive species. Further to this, it recognises that the most serious social impacts from invasive or pest species are likely to result from economic impacts on regions highly dependent on specific plants or animals that may be attacked by a new disease or pest. Their review noted that "water quality and decline and reduction in native fish species leads to social impacts through reduced recreational fishing opportunities, limits on other water recreational activities, and tourism". In addition, aquatic vertebrates were analysed based on their economic, environmental and social impacts. The methodology considered rough estimates of current costs of controlling the species, losses to commercial industry, agricultural impact, tourism, recreational fishing, water quality, and negative impacts on other native fish species. Among other species, it found that gambusia, redfin perch and tench all have impacts on native fish and other aquatic species.

Key findings from the literature review

The literature review identified a number of key findings that have informed the development of the Social Impact Assessment Tool. These are identified below:

- Potential social impacts from invasive species include: damage to ecosystems; declines in local economies, including numerous industries, such as fisheries, eco- tourism and electricity production and employment opportunities; and loss of recreational fishing opportunities, amenity values.
- Commercial harvesting of invasive or pest species may have positive economic impacts for local economies and employment; however it is necessary to balance these impacts against management costs, and decline in social wellbeing due to loss of recreational activity opportunities, as well as environmental amenity. Previous studies have highlighted that these are difficult to quantify.

- It is important to balance and evaluate the full spectrum of views in identifying the most appropriate action for managing invasive species. This is demonstrated where the control of an invasive species may only be sought by a particular sector of the community rather than universally.
- Consideration needs to be given to the short and long term effects of species introduction, to ensure that future generations are not disadvantaged by current decisions.
- Impacts need to be measured by the level of threat, from low to major risk. Where the level of impact is unknown it is important to ensure that this knowledge gap is identified. An ‘impact unknown’ category is as significant to the SIA as a major impact because it highlights the need for further research.
- Impacts on indigenous cultural heritage are often ignored during risk assessment processes, despite the important links maintained by indigenous peoples to the natural environment. Therefore indigenous values, in terms of social impacts of fish species affecting traditional cultural norms, should be included in the SIA.
- The availability of quantitative data (population dynamics, employment, recreational fishing data, recreational amenity/property values) is important when measuring the social and economic impacts of invasive species.
- Eastern gambusia, redfin perch, tench, and roach are all perceived to pose some threat to recreational fishing, water quality and the economy.
- Public perceptions of invasive species can be measured by ascertaining, first, a community’s awareness of the species’ distribution and impacts, and consequently, whether they believe these impacts, overall, to be positive, negative, or benign.

6.4 Social impact assessment framework

The International Association for Impact Assessment (IAIA) provides useful guidance on the role and scope of SIAs.

‘A Social Impact Assessment (SIA) is analysing, monitoring and managing the social consequences of development. Social impact assessment includes the processes of analysing, monitoring and managing the intended and unintended social consequences, both positive and negative, of planned interventions (policies, programs, plans, projects) and any social change processes invoked by those interventions’.

The steps involved in undertaking the SIA for pest fish species in Australia are shown in Figure 6.1[4] below and include the monitoring and review process of each social

[4] Based on the model developed by Howard (2003), Collaborations: Planning with your community, South Melbourne ©2004

impact and the ongoing role of the community consultation strategy that are needed to measure the ongoing impacts of the fish over time.

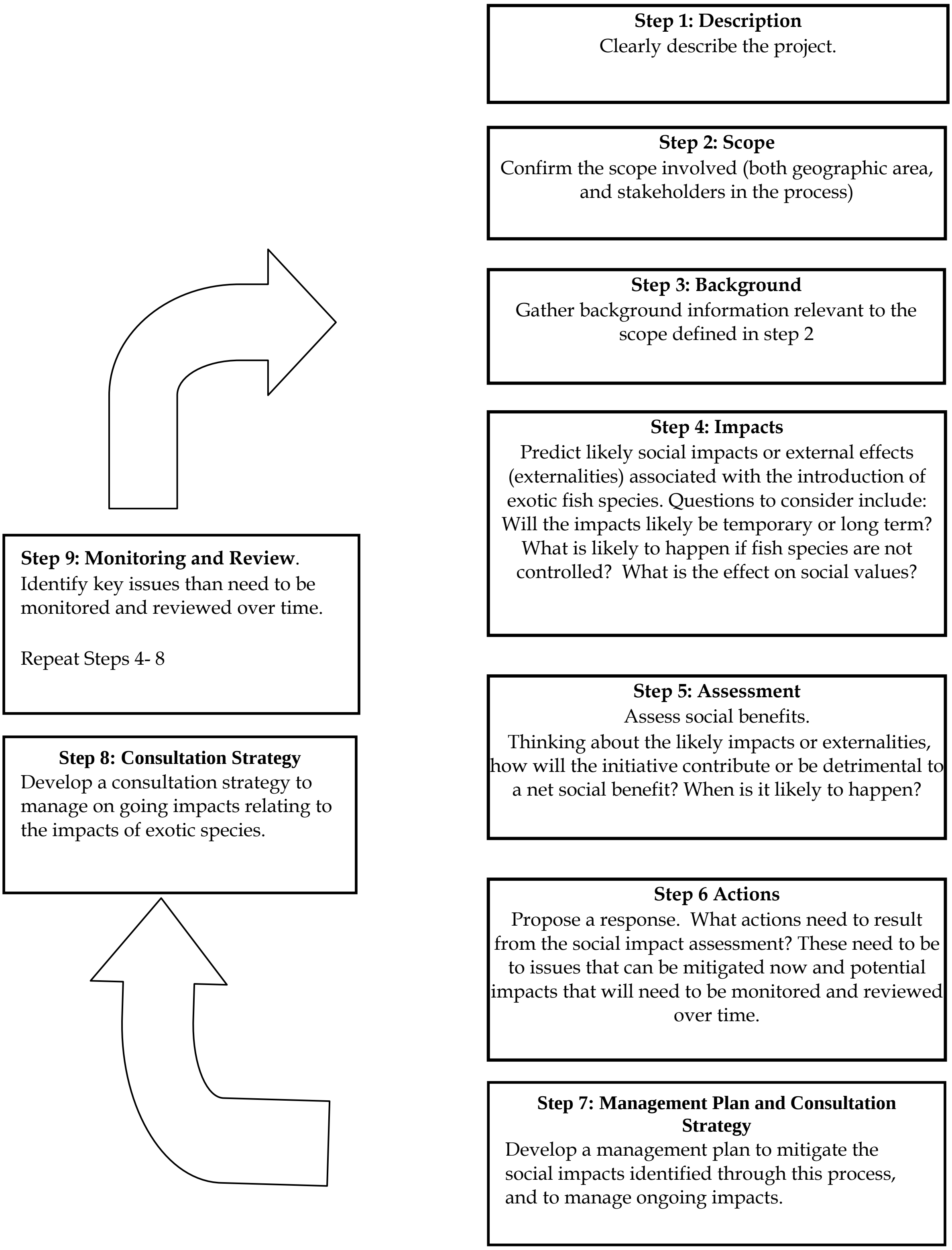


Figure 6.1: Impact assessment framework flow chart.

Key areas of social value relevant to this SIA framework include people's:

- Way of life - how they live, work, play and interact with one another on a day to day basis.
- Culture - shared beliefs, customs, values, languages or dialects.
- Community - social cohesion, stability, character, services and facilities.
- Political systems – the extent to which people are able to participate in decisions that affect their lives, the level of 'democratisation' that is taking place, and the resources provided for this purpose.
- Environment – the quality of water people use; the availability and quality of the food they eat; the level of hazardous risk; their physical safety, and their access to and control over resources.
- Health and wellbeing – health is a state of complete physical, mental, social and spiritual wellbeing and not merely the absence of disease or infirmity.
- Personal and property rights – particularly whether people are economically affected, or experience personal disadvantage which may include a violation of their civil liberties.
- Fears and aspirations – perceptions about safety, fears about the future of the community and aspirations for the future and the future of children.

Based on the literature review and emerging likely impacts of each pest fish species from a socio-economic perspective, the main socio-economic and cultural attributes appropriate for this SIA framework were identified and are described below:

Way of life: Alien pest fish are likely to affect people's way of life due to impacts on recreational fishing and commercial fishing industries. Some of the measurable likely impacts include impacts on recreational opportunities and impacts on employment

Culture and community: Perceptions of alien pest fish species are likely to vary between different societal groups. This will shape people's attitudes towards alien fish and whether they view them positively or negatively. For example, indigenous communities with strong values associated with native fish for cultural and spiritual reasons, may feel less positive about pest fish than others. Similarly, different groups in the community will perceive alien species differently depending on whether they come from a conservation, scientific or industry/recreational perspective. Some of the measurable likely impacts include those on indigenous cultural heritage and beliefs and impacts on community values.

Environment: Because pest fish species may affect people's interaction with their environment, it may be appropriate for public participation to be incorporated into decision making for future management.

Health and well being: Alien pest fish species may impact on the natural ecological balance (see Chapter 4), leading to decreased amenity values, recreational opportunities and potential health issues due to parasites, disease or pathogens. Not only can alien fish spread disease and pathogens, but through reducing native fish populations, and damaging the environment, they can impact on opportunities to participate in recreational fishing which is an important activity to enhance both physical and mental wellbeing for some people. Impacts on health and wellbeing due to potential for spreading of parasites and disease (e.g., redfin perch) also need to be considered along with impacts on biological communities (i.e., loss of native species diversity).

Personal and property rights: Where alien pest fish negatively affect fishing industries, social and economic impacts are likely to include loss of income and employment. Property values may also be impacted if private dams or access to water bodies are affected by invasive species. Impacts on amenity values (i.e., fisheries and water quality) also need to be considered.

Fears and aspirations: Alien pest fish species might induce feelings of fear and uncertainty where there is a possibility that they will affect the environment, native fish populations, fishing industries and associated livelihoods. Impacts on ecosystem values may also be a concern and include loss of or decline in native species populations through predation, competition, parasites/pathogens, disease, decline in growth rates of native populations, alteration of behaviour in native species populations.

Information is also required on the risk posed by the impact of pest fish on each area of social value. A risk assessment framework was therefore developed to help quantify this. In this framework, a classification of risk has been applied to each area of social concern to determine the "likelihood" of pest fish impacting upon it. The matrix in Table 6.1 outlines the criteria that were used to classify the level of risk.

Table 6.1: Risk Matrix[5].

Probability of occurrence of an impact	Potential impact on social value		
	Low (No or minimal adverse social impact)	**Medium** (Moderate social impact)	**High** (Significant damage to community)
Low (Event may occur at some stage)	Low	Low	Medium
Medium (Event may occur occasionally)	Low	Medium	High
High (Event may occur regularly)	Medium	High	High

Where an impact has a low or minimal adverse social impact, but is highly likely to occur, that particular species or impact will have an overall medium risk. Likewise, where an impact may be considered as high risk, but the likelihood of its occurrence is low, the overall risk category is lowered to medium.

Short and long term risks were defined as:

- Short-term: May impact on immediate access opportunities for fishing or enjoyment of freshwater environments, including human health.
- Long-term: May impact on ecosystem health and sustainability of recreational fishing industry.

6.5 Results of stakeholder consultation

Responses from meetings and phone interviews were collated in tabular form using the SIA framework. The information gathered in this process was used to provide anecdotal feedback on the social values identified in the literature review. Some participants referred to quantitative and qualitative data results (i.e., surveys and studies) - these are noted throughout the findings. This section provides an account of the main findings.

6.5.1 Impacts on way of life

Way of life is how people live, work, play and interact with one another on a day to day basis. This includes personal and property rights – particularly whether people are economically affected, or experience personal disadvantage which may include a violation of their civil liberties.

[5] Developed by Michelle Howard, Collaborations: Planning with your community, South Melbourne ©2004 based on the internationally accepted model for assessing risk

As a recreational opportunity, fishing is highly valued in Australia. Given the number of people identified who are involved in fishing activities, it was assumed at the outset of this project that a number of the fish species selected for this study would be valued by the community for recreational purposes. As a flow-on effect, it was also assumed that the economic benefits would be valuable in terms of local fishing industry, including equipment, gear, bait, tourism opportunities, boating and accommodation in local towns.

Impacts on personal property rights also have implications for overall wellbeing. It is therefore important to understand from a social perspective how the introduced fish species impact on personal and property rights. This means how they may impact on personal monetary wealth including property values and amenity values. This section provides an overview of the likely impacts both positive and negative that each species may have on people's way of life and day to day activities.

6.5.2 Impacts on recreation

Workshop participants and telephone interviewees' perceptions of each species of introduced fish varied amongst stakeholder groups, and further differed between the state jurisdictions.

A representative from the Victorian peak fishing body expressed the view that in Victoria, redfin perch is considered highly valuable for recreational angling throughout inland waterways. The representative also suggested that redfin is very popular because of its resilience to water temperatures and tolerance for different environments. In particular it was mentioned that due to their abundance in Victorian waterways, redfin provide an opportunity for young people to experience the excitement of catching a fish and to learn about fishing.

The coarse angling method is to catch and release fish and is historically linked to carp, redfin, tench and roach. Because coarse fishers do not catch fish to eat, the value of the fish caught does not depend on its table qualities. Coarse angling was born out of poaching in the days of the aristocracy; in order to avoid being caught, fishers developed methods that enabled them to catch fish quickly. Nowadays, the term "coarse" is considered by some anglers to be old-fashioned and derogatory (in Victoria in particular, redfin is no longer known as a coarse fish), however, the method of fishing is still strongly supported. At the present time, in Australia, coarse fishers enjoy the experience of catching fish and practicing a skilled and independent sport that appeals to independent minds.

In NSW and SA, redfin perch is not such a popular fishing target. Coarse anglers in NSW mentioned that any species of fish can be caught for the sport, however, it is more desirable for anglers to have a variety of fish species to target. It was therefore stated that if any of the introduced species were to overpopulate and reduce diversity within the waterways, from their perspective it would reduce the overall recreational fishing experience. There was some concern over the resilience of redfin in NSW, where populations can become very large and out-compete other fish for resources and habitat. Redfin is also a predator, is known to be aggressive, and can invade other fish species' territory. In this respect, the presence of redfin perch can affect the development of other more desirable sports fisheries. For example, perch feed on fingerling trout and therefore constrain the development of trout fisheries, forcing the stocking of more expensive yearling fish (Molony et al. 2004).

Although the use of live fish as bait is prohibited in New South Wales, redfin perch can be used for live bait to catch other fish species in other states and therefore have value other than as a target species. In addition, roach have also been used to catch redfin.

Other than redfin, tench was introduced to NSW in the 1940s, both as a food source and for the sport of coarse angling. According to coarse anglers consulted in NSW, tench is more highly regarded as it has a small population base in the state and is therefore more prized. Similarly, according to fishers in NSW, roach have minor populations in NSW, although no formal populations have been recorded. This may be due to their habitat preference for cooler water temperatures. The majority of people consulted were of the opinion that roach and tench were fairly benign in terms of their social impacts with regard to recreational opportunities. Both were introduced in Australia for acclimatisation[6] and they quickly filled a niche. However, they tend to only inhabit cooler climates in the southern states.

gambusia on the other hand was perceived by most stakeholders as a "nuisance" fish. All participants seemed to agree that is was no more effective than native species for mosquito control and that it has no value among the community for recreational fishing or for its table qualities. In fact, coarse angling clubs in NSW have banned gambusia from their competitions, as the size of the species is small and it can easily swim into a net, therefore increasing overall catch numbers; recording gambusia is considered a form of cheating.

Little was known about the likely social impacts of gobies on recreational values. A comment was made that in Port Phillip Bay introduced gobies could impact on the diving experience if they became a pest species like the crown of thorns sea star, however this was purely speculation. Coarse anglers view gobies as "just another fish to catch," so do not perceive them to be adversely impacting on the recreational experience; however they are relatively small and therefore not specifically targeted.

[6] DPI Victoria - Fisheries

6.5.3 Impacts on local economies

Recreational fishing in Australia is a highly valued sport and pastime for a large proportion of the population. The economic flow-on effects of inland recreational fishing are beneficial for local towns, tourism, fishing associated industries and for local employment, and contribute approximately $460 million to the national economy each year.

According to those consulted in each state, the economic value of fishing will be high because of its social values. In particular, coarse anglers specifically target fish included in this study, redfin perch, roach and tench. Coarse angling is an expensive sport, which requires different gear for different conditions. The value of the industry includes fishing gear and fishing competitions that bring people into towns (such as the Horsham Redfin Championship), providing economic and social benefits to local residents and local industries. It also creates regional tourism opportunities.

In spite of the positive impacts that fishing provides to the economy, there are also a number of adverse impacts resulting from illegal fish stocking and the corresponding costs of research into control and management of introduced species in the waterways. In Victoria, NSW and SA it is illegal to stock redfin perch. Despite this, anecdotal evidence from fishers and other stakeholders suggests that redfin, roach and tench have all been illegally introduced into waterways. Fishers, who have recorded catches in locations that the species have not previously inhabited, have proven this.

Other comments from consultation include the perceived detrimental economic impact on fishers that spend money on bait, only to have it 'wasted' when they catch fish that are illegal in competitions (such as gambusia) or protected native species that they have to throw back. This is a minor issue.

Although redfin perch is a valuable species for anglers, it is also predatory and biologically very resilient. Similarly, gambusia poses a threat to native fish species and has been known to be aggressive and nip the fins of other fish (see Chapter 4). According to stakeholders, fisheries departments have spent time and money in researching methods of controlling introduced fish species such as redfin and gambusia.

New South Wales DPI noted that rotenone is a registered fish poison in NSW. However, it is only appropriate to use rotenone in specific circumstances such as isolated water bodies. Because of this limitation, it has not been successful in reducing overall gambusia or redfin perch numbers. Gambusia, more than any other fish species, is expensive to control and eradication has proven next to impossible. Attempts have been made in the past and have only been successful in isolated bodies of water (see chapter 8)

6.5.4 Impacts on property and amenity values

A number of angling clubs use private dams for fishing competitions and, if undesirable fish species were to be introduced to these isolated bodies of water, this may adversely impact on the value of that fishing resource. Likewise the presence of a good fishery on private land or nearby would impact positively on land values and property values.

The South Australian Freshwater Angler's Association conducts fishing competitions and opportunities for recreational freshwater fishing on private properties of club members. The association has a conservation policy to ensure that fish stocks are sustained and preserved for all to enjoy. One of their rules for members to observe is not to introduce gambusia or redfin perch. It is also against the law in SA to introduce these species into new areas (see Chapter 9).

In order to maintain recreational fisheries, NSW DPI have recently completed a Freshwater Fish Stocking Strategy. This included an Environmental Impact Statement to justify why fish stocking is necessary.

6.5.5 Impacts on health and well being

Health is a state of complete physical, mental, social and spiritual wellbeing and not merely the absence of disease or infirmity. A reduction in overall health and well being within a community has the potential to be detrimental to happiness and economic productivity. Any impacts that introduced fish species may have on people's health and wellbeing therefore need to be considered in order to provide a balanced recommendation for future management options.

The benefits and values of recreational fishing are large. Fishing enhances people's ability to experience natural environments, and contributes to personal health and wellbeing through a sense of community connectedness and enjoyment. According to some stakeholders, recreational fishing brings life into communities. Stakeholder discussions mentioned the importance of recreational fishing as a stress-relieving emotional outlet and referenced the recent drought and corresponding financial strain on rural farming life. To demonstrate this, the participant explained that there is a high rate of depression and suicides among farmers facing debt and hardship and that being able to experience recreational retreat through fishing was a fundamental source of increased wellbeing. Fishing is important in these communities for relaxation and improved mental health. It helps reduce the contemplation of suicide.

Redfin perch, particularly in Victoria, have stable populations and provide a year-round target for fishing, providing sustainable recreational opportunities. In some cases, redfin are more highly valued than trout. None of the other species were mentioned in discussions as providing any real benefit to health and wellbeing.

The National Recreational and Indigenous Fishing Survey (2002) reinforced these views and demonstrated that there are perceived psychological, environmental and social benefits as well as sporting aspects as motives for fishing. These all relate to the issues identified by stakeholders in this study. It further identified that the non-catch related motives (to relax and unwind, to be outdoors, for solitude, to be with friends and family) rated more highly than catch related motives for fishing (for food, for competitions and sport).

6.5.6 Impacts on culture and community

Culture includes shared beliefs, customs, values, languages or dialects; community-social cohesion, stability, character, services and facilities. Building upon the values of recreational fishing in Australia it is equally important to balance the views from a cultural and community perspective and how invasive or introduced species are perceived within the wider community.

Indigenous cultural heritage and beliefs: Consultation with stakeholders as well as the literature review identified that native fish species are important to Aboriginal people. In particular, the cultural value of Murray cod derives from the part it plays in a Dreamtime story. Fishing also provides indigenous people with a food source and connections with ceremonial occasions, trade and barter (NSW 2002).

Although some introduced species may provide a source of food, particularly redfin perch, other introduced species in this review are not commonly eaten, and have the potential to compete for resources with other more highly regarded native fish species. Declining native fish populations have resulted in the closure of fishing seasons to allow the species to reproduce to population sizes suitable for sustainable harvesting. Although native fish species are affected by many factors, pest fish species are likely to be a contributing factor in reducing numbers of some native fish species such as Murray cod and Murray crayfish. The National Recreational and Indigenous Fishing Survey (Henry and Lyle 2003) identified a number of values relevant to the cultural links of indigenous people in Australia in relation to fish species. The survey included indigenous communities from Queensland, Western Australia, and Northern Territory and it demonstrated the potential translocation of the values of recreational and cultural fishing practices to other states. It recognised that the "Australian landscape is a product of the economic modes of subsistence of indigenous people and is important in both cultural and religious terms." In addition, the survey found that fishing is an "invaluable component of the cultural lifestyle of indigenous people and is connected to the traditional responsibilities of land management and kinship." Further to this, the social value of food collecting is important in maintaining social networks through the sharing of gathered food.

Indigenous people have traditionally managed fisheries through knowledge gained over thousands of years, experimentation, mythology and lore. They developed sustainable fishing methods, and excluded certain areas from fishing for alternate seasons. Current practices and fishing activities do not distinguish between indigenous cultural and community activities and recreational or commercial fishing. This means that indigenous communities must adhere to relevant bag size limits, closed areas and licensing arrangements. Given some species of native fish are threatened and are protected from fishing, this has implications for traditional fishing for cultural purposes. A broad link can thus be made between the presence of introduced pest species, which has potential implications for traditional fishing opportunities, and the need for protection of native fish.

Understanding the importance of fisheries access to indigenous communities, the NSW Indigenous Fisheries Strategy (NSW Fisheries 2002) aims to support the ongoing exemption of traditional cultural fishing from recreational fishing licensing schemes. An overarching aim of the Strategy was the desire to preserve fisheries for the future; a desire shared by the general recreational fishing community in Australia. This was also the case found in consultation with indigenous communities in the Wet Tropics Bioregion workshops (Larsen and Parnell 2002), where water quality was identified as important for the maintenance of cultural resources (such as fish species) and practices.

Community values: Apart from fishers and those involved in native fish protection, there appears to be little community awareness of the value (or lack thereof) of introduced fish species in Australian inland waters. A comment was made by more than one stakeholder, however, that introduced species were appropriate in controlled environments, or isolated bodies of water that posed no threat of flooding into a river or stream system.

Representatives from NSW DPI commented that there is a need for enhanced awareness of fish stocking. From a scientific perspective, they perceive that there is not enough community awareness of the impacts of redfin perch in NSW. Community education appears to be not sufficiently widespread and the value of native fish in Australian waters needs to be more effectively communicated to the wider community.

The value of these introduced fish species for their table qualities varies. Redfin perch are valued for their taste and are known to be a good table fish. Roach is not widely eaten, however there is a perception that the Chinese community may value this species and have special methods of preparing the fish for eating. Gambusia is not an edible fish, while tench and the gobies are not valued as table fish.

In order to effectively manage introduced species and reduce their potential to impact on the environment and ecosystems, public education programs are necessary tools. According to Lintermans (2004), "unless people are aware of the potential damage to aquatic environments from alien species, the number and spread of alien species will continue to rise. Education programs are long term investments that require a sustained and ongoing commitment, but unless there is community understanding and support for alien management programs, they are doomed to failure." Similarly, Webb (2006) states "Community motivation to participate or cooperate with conservation strategies may be affected by adverse perceptions of fisheries management either doing too little or intervening too late to resolve an important environmental problem, the use of control measures which conflict with community values or have adverse impacts on other aquatic flora and fauna, and frustration of recreational fishers prevented access to waters undergoing control measures." Koehn and MacKenzie (2004) also stress the importance of community education in ensuring the effectiveness of pest management, noting that education decreases the risk of community members spreading alien species, and increases the likelihood of public participation in reporting infestations.

6.5.7 Impacts on environmental sustainability

Environment is the quality of water people use; the availability and quality of the food they eat; the level of hazardous risk; their physical safety, and their access to and control over resources. Stakeholder comments revealed that redfin perch and gambusia in particular benefit from water regulation because it creates a more suitable habitat. There is also some speculation that both redfin and gambusia could be linked to blue green algal blooms. Given the detrimental health and well being impacts associated with blue green algal outbreaks, this has the potential to cause a number of adverse impacts to nearby residents, fishers and water users.

6.5.8 Impacts on environmental values

In NSW, redfin perch are perceived to impact on Macquarie perch and silver perch populations. This has the potential to impact on recreational opportunities as it may result in protection of these native fish, rendering the fisheries untargetable for anglers. Other likely impacts on species diversity include the redfin perch's susceptibility to the epizootic haematopoietic necrosis virus which, according to researchers, has been known to kill Macquarie perch. Due to population decline, Macquarie perch have been stocked in the Murray Darling Basin. The spread of fish-borne disease has social implications in that as well as impacting on recreational fishing opportunities for native fish, it can also reduce diversity in the waterways.

Gambusia on the other hand has been implicated in the decline of a number of threatened species including particular species of native frogs (see chapter 4). Most stakeholders agree that gambusia does not have any redeeming features. There is universal acceptance that the reason for its introduction to control mosquito populations was a "failed experiment" and that it is less effective at controlling mosquitoes than some native fish species such as galaxiids, gudgeons or rainbowfish. Due to their aggressive nature, gambusia are not a welcome fish species in any water body. They are perceived to have more of an impact in confined waterbodies, and in particular where native ecosystems are already stressed.

Although not a lot of information was known about tench, a comment was made that the species was once known as the ‘doctor fish’ because of its reputed healing properties. According to fishers, other species of fish, when sick, have been known to rub their fins against the mucous that tench produce, for a healing effect. This has the potential to be studied for its positive impact on native fish species - and also whether its healing properties are transferrable to people.

6.5.9 Fears and aspirations related to impacts

Fears are the perceptions people hold about their safety and the future viability of their community. Aspirations are their hopes for the future and the future of their children. Fears and aspirations and perceptions of impacts are just as important as actual impacts, as they comprise a person’s perspective on life relative to their environment. In spite of fish anglers commenting on the value of their fisheries, there was also a common belief that it was just as important to maintain species diversity in the waterways including native fish populations. Given that fishing relies on healthy and diverse waterways, there was considerable concern for the environment. A comment was made that most fishers would prefer for an introduced species to be eradicated if it was proven that it caused detrimental effects on ecosystems, as the overall health of an ecosystem is more important than the presence of one species of fish.

Redfin perch is a case in point. They are highly popular as a fishing target; however they also have a reproductive strategy that leads to high abundance and overconsumption of shared resources. Redfin are very resilient, and become stunted when population numbers increase so that they can still occupy the same niche and therefore out-compete native fish. This has potential impacts on native fish and on recreational opportunities because the number of legal sized fish able to be caught decreases as redfin abundance increases.

Many of the key policy documents (see Chapter 9) discuss enhancing community awareness of the issues associated with pest species. This is the key to ensuring that fears and aspirations are relieved and realised, respectively, in relation to recreational fishing and conservation of native ecosystems.

6.6 Summary of socio-economic and cultural impacts

The SIA framework based on both the socio-cultural attributes likely to be affected by alien pest fish and the risk levels in Table 6.1 was used to assess the potential socio-economic impacts of the alien fish species reviewed here. The results are shown in Table 6.2. The risk levels have been generalised across the whole study area. Location specific assessment needs to be undertaken where each species is found to be prevalent to provide a more accurate assessment. This assessment has been undertaken as a guide only.

Table 6.2: Summary of SIA findings.

NIWA Australia

			SHORT-TERM	Risk Level	LONG-TERM	Risk Level		
Possible Impact	**Potential Issues**	**Data Source**	**Likely Short-Term Impact to Measure**		**Likely Long-Term Impact to Measure**		**Recommendations**	**Key Stakeholders to Consider**
Way of life	Access to recreational opportunities	▪ Numbers of illegal activities – calls to Illegal Fishing Hotline 133 474 (DPI (NSW) ▪ National Recreational and Indigenous Fisheries Survey ▪ Coarse anglers fish records (databases) ▪ Public fishers database DPI ▪ Inland angling waters of Victoria www.dpi.vic..gov.au/inlandangling	Increased access to a variety of fish species with presence of introduced fish and therefore opportunity to catch a fish. Likelihood of gambusia (a species not valued for recreational purposes) impacting on other recreational fish and therefore reducing target fish species populations.	Medium	Some reduced recreational opportunities due to likelihood of redfin stunting their growth in large populations (therefore out-competing other fish and in their small size being under regulation size for acceptable catch). Illegal stocking of redfin, roach, and tench increasing competition for habitat, and reducing species diversity and therefore overall health of ecosystem. Increased number of river or water body closures ceasing recreational fishing due to the need to protect native fish breeding.	Medium	-Registration of illegal activities by species and state -Continuous updating of National Recreational Fishing Survey sent to all recreational fishermen - Communication between fishers and fisheries departments on fish caught for recording purposes.	Representatives from each state and territory fisheries department (DPI or DOI). Peak fishing bodies i.e., VR fish, NSW Council of Coarse Anglers, South Australian Freshwater Anglers Association (SAAFWA) Australian Federation of Coarse Anglers Native Fish Australia Conservation Organisations
	Impact on local economies and impacts on tourism	▪ Literature review ▪ Stakeholder consultation ▪ National Recreational and Indigenous Fisheries Survey ▪ Tourism visitation and expenditure in local towns related to fishing (e.g., Horsham Redfin Championship).	Increased local revenue from fishing competitions bringing people into nearby towns and flow-on employment activities. Increased community awareness through conservation programs.	Low	If found that ecological impacts of fish species are detrimental to the sustainability of freshwater ecosystems, fishing opportunities may also be detrimentally affected in the longer term, as would local dependent economies. Conservation and associated employment will positively benefit from creating community awareness and regulation.	Medium	Research into the available data or the development of an economic survey to calculate value of recreational fishing in the community for local economies.	Representatives from industry associated with freshwater fishing and coarse angling. Peak fishing bodies Local tourism authorities
	Impact on local amenity values	▪ Literature review ▪ Stakeholder consultation ▪ Numbers of private properties that host fishing competitions ▪ Water watch program data for water quality and biodiversity counts	Increase in local property values or personal wealth of properties where fishing competitions occur or where valuable fishing opportunities exist. Increased awareness of sustainable fishing outcomes and sustainable fishing targets.	Low	Increased awareness for sustainable fishing opportunities and shift to sustainable targets and practices. Desire to maintain valuable fishing opportunities likely to lead to improved fishing practices.	Low	Research into what makes a landscape/waterway valuable. Continue to monitor, or commence monitoring private fishing practices and fish stocking activities. Enhance awareness among fishing community of sustainable fishing practices.	Coarse angling clubs that conduct fishing competitions on their properties. Local residents and 'friends of waterways' groups. Waterwatch officers Native Fish Australia

			SHORT-TERM	Risk Level	LONG-TERM	Risk Level		
Possible Impact	**Potential Issues**	**Data Source**	**Likely Short-Term Impact to Measure**		**Likely Long-Term Impact to Measure**		**Recommendations**	**Key Stakeholders to Consider**
	Impact of/on tourism	▪ Access to tourism ▪ Employment opportunities ▪ ABS tourism indicators and expenditure on recreational fishing nationally ▪ Ecotourism operators	Increase in tourism dollars through attraction of valuable fishing locations, including flow-on effects of overnight stays and restaurants, shops etc.	Low	Increase in local employment opportunities should fishing tours and supporting opportunities develop around particular fishing festivals, competitions, or touring routes.	Medium	Research into the sustainable development of ecotourism opportunities for fishing.	Ecotourism operators Local tourism authorities Peak fishing bodies Local fishing clubs
Health and wellbeing	Impact to personal health and wellbeing	▪ Literature review ▪ Stakeholder consultation ▪ Beyond Blue - statistics of people living in rural areas with depression ▪ DSE – Regional Atlas health and well being data including mental health statistics in rural communities	Reduced likelihood of depression in rural areas due to recreational outlet of fishing.	Medium	Continued opportunities for fishing will result in improved overall community wellbeing. Potential for introduced species to reduce variety of fishing opportunities and therefore reduce the fishing experience. Increased number of river or water body closures ceasing recreational fishing due to the need to protect native fish breeding.	Medium	On-going monitoring of community attitudes through the National Recreational and Indigenous Fishing Survey.	Peak fishing bodies Recreational fishing clubs Community health providers Representatives from Department of Planning and Community Development (DPCD)
Culture and environment	Impact on indigenous cultural heritage and beliefs	▪ Literature review ▪ Stakeholder consultation ▪ National Recreational and Indigenous Fishing Survey ▪ Local Aboriginal Co-operatives	Potential for reduced access to native species leading to reduced ability to practice traditions and cultural practices.	High	Reduced access to highly valued traditional or totem native fish where introduced species out-compete for habitat and resources. Increased number of river or water body closures ceasing recreational fishing due to the need to protect native fish breeding.	High	On-going monitoring of community attitudes through the National Recreational and Indigenous Fishing Survey. Develop a survey that addresses aboriginal cultural heritage beliefs regarding introduced fish species.	Representatives from local aboriginal co-operatives Representatives from fisheries departments Murray Darling Basin Commission Aboriginal Representative
	Impact on community values	▪ Literature review ▪ Stakeholder consultation ▪ Gregory's Guide to Fishing ▪ Coarse anglers' records	Increased concern among native redfin perch anglers that redfin may move into its current niche. Increased concern over the presence of gambusia in the waterways	Medium	Increased awareness of the need to stop illegal stocking of introduced fish species (in particular, gambusia and redfin)	Medium	On-going monitoring of community attitudes through the National Recreational and Indigenous Fishing Survey.	Local community groups, conservation and land-care action groups Native Fish Australia

			SHORT-TERM	Risk Level	LONG-TERM	Risk Level		
Possible Impact	**Potential Issues**	**Data Source**	**Likely Short-Term Impact to Measure**		**Likely Long-Term Impact to Measure**		**Recommendations**	**Key Stakeholders to Consider**
Fears and aspirations	Impact on native species population	▪ Literature review ▪ Stakeholder consultation ▪ Native Recreational and Indigenous Fishing Survey	Fear that introduced fish species may impact on the number and variety of fish in water ways. Fear of increased occurrence of competition for resources, fin nipping and dispersal of pathogens or diseases.	Medium	Increased awareness of the intrinsic value of the environment and the need to ensure it is protected.	Medium	Support and monitoring of key findings from the Action Statement for Introduction of Live Fish into Waters Within Victorian Catchments.	Local community groups Coarse angling clubs Native fish angling groups Native Fish Australia Australian Federation of Coarse Anglers

The preliminary social impact assessment of the alien fish species reviewed has resulted in the following key findings.

- Redfin perch are highly valued in Victoria by recreational anglers, and are of medium value in NSW and SA.
- Gambusia are not valued by any sector of the community.
- Roach and tench are coarse fish angling targets in Victoria and less so in NSW.
- The gobies have a perceived benign social impact in that not much is known about them and community awareness of their presence is low.
- Redfin perch, roach, tench, and gambusia are predators and are likely to compete for resources of native fish species and therefore likely to impact on native species populations.
- Redfin perch are known to carry a virus and are likely to pass it on to other fish species.

In addition, the review of literature has identified the following knowledge gaps:

- Specific or quantitative measures on the social impacts of invasive species and the seriousness of these impacts are lacking.
- There is little or no assessment of social and economic impacts on indigenous and non-indigenous communities.
- Little is known about the yellowfin and streaked goby, which indicates that further research will need to be undertaken in order to develop baseline data to assess ongoing and potential future impacts.
- Risk assessments have rarely been undertaken prior to the release of an introduced species into a non-native environment, therefore no data exists from the time of introduction to measure change over time of population dynamics and impacts from establishment.

The social impact assessment shows that:

- Location-specific assessment needs to be undertaken where each species is found to be prevalent to provide a more accurate assessment.
- Most negative short- and long-term impacts relate to increasing populations of fish that are considered pests, and the effects such increases will have on the ecological and recreational potential of waterways.
- Most positive long-term impacts identified in this assessment relate to improving access to redfin recreational fishing, and the benefits this will offer to nearby townships and the region.
- Management plans should be put in place to monitor social issues identified in this report and likely to arise from any actions based on it. These should include strategies and actions for both community and stakeholder consultation.
- Relevant authorities should ensure that, where appropriate, contract documentation relating to the management of the fish species reviewed in this report includes a requirement for monitoring and review of social impacts.
- It is also important to involve the community in monitoring activities, to create a sense of ownership of the pest fish issue and an improved understanding of the difficulties in managing introduced species.

The potential issues and their likely short or long term social impacts are outlined in Table 6.3.

Table 6.3: Perceived short- and long-term social impacts.

Possible Impact	**Potential Issues**	**SHORT-TERM** **Perceived Social Impacts of Pest Fish**	**Positive or Negative Impact**	**LONG-TERM** **Perceived Social Impacts of Pest Fish**	**Positive or Negative Impact**
		Likely Short-Term Impact to Measure		**Likely Long-Term Impact to Measure**	
Way of life	Access to recreational opportunities	Increased access to a greater variety of fish species with presence of introduced fish and therefore opportunity to catch a fish. Likelihood of gambusia impacting on other recreational fish and therefore reducing target fish species populations.	Positive Negative	Some reduced recreational opportunities due to likelihood of redfin stunting their growth in large populations (therefore out-competing other fish and, in their small size, being under regulation size for acceptable catch). Illegal stocking of redfin, roach, and tench increasing competition for habitat, and reducing species diversity and therefore overall health of ecosystem.	Negative Negative
	Impact on local economies Impacts on tourism	Increased local revenue from fishing competitions bringing people into nearby towns and flow on employment activities.	Positive	If found that ecological impacts of fish species are detrimental to the sustainability of freshwater ecosystems, fishing opportunities may also be detrimentally affected in the longer term and therefore local dependent economies. Conservation and associated employment will positively benefit from creating community awareness and regulation.	Negative Negative
	Impact on local amenity values	Increase in local property values or personal wealth of properties where fishing competitions occur or where valuable fishing opportunities exist.	Positive	Fishers' desire to maintain valuable fishing opportunities likely to lead to improved fishing practices, for example preferring native fish species, and those species that do not impact on the diversity of fish species, or quality of water course.	Positive

Possible Impact	Potential Issues	SHORT-TERM Perceived Social Impacts of Pest Fish	Positive or Negative Impact	LONG-TERM Perceived Social Impacts of Pest Fish	Positive or Negative Impact
		Likely Short-Term Impact to Measure		**Likely Long-Term Impact to Measure**	
	Impact of/on tourism	Increase in tourism dollars through attraction of valuable fishing locations, including flow-on effects of overnight stays and restaurants, shops etc.	Positive	Increase in local employment opportunities should fishing tours and supporting opportunities develop around particular fishing festivals, competitions, or touring routes.	Positive
Health and wellbeing	Impact to personal health and wellbeing	Reduced likelihood of depression in rural areas due to recreational outlet of fishing.	Positive	Continued opportunities for fishing will result in improved overall community wellbeing. Potential for introduced species to reduce variety of fishing opportunities and therefore reduce the fishing experience.	Positive Negative
Culture and environment	Impact on indigenous cultural heritage and beliefs	Reduced access to native species leading to reduced ability to practice traditions and cultural practices.	Negative	Reduced access to highly valued 'traditional' native fish where introduced species out-compete for habitat and resources. Increased number of river or waterbody closures affecting recreational fishing due to the need to protect native fish breeding.	Negative Negative
	Impact on community values	Increased concern among native perch anglers that redfin may move into its current niche. Increased concern over the presence of Gambusia in the waterways.	Negative Negative	Increased awareness of the need to stop illegal stocking of introduced fish species (in particular, gambusia and redfin).	Positive
Fears and aspirations	Impact on native species population	Fear that introduced fish species may impact on the number and variety of fish in waterways. Increased occurrence of competition for resources.	Negative Negative	Increased awareness of the intrinsic value of the environment and the need to ensure it is protected from pest fish.	Negative

6.7 Recommendations

This work has developed a framework for an SIA assessment of the impacts and control of alien fish species in Australia. A preliminary assessment has been undertaken on a range of species and has been generalized to incorporate study areas throughout Australia across a range of locations. The next step is to undertake a detailed assessment and to select a location and species to test the SIA framework. Once the location and species have been selected, the eight issues identified in Table 6.4 should be investigated further as outlined in the SIA flow chart (Figure 6.1).

Table 6.4: Steps for a more detailed, location and species-specific assessment.

Step (Refer to Figure 6.1)	Action
Step 1: Describe the project	• Select a site and species based on findings from this report.
Step 2: Scope the project	• Confirm the site, species and key stakeholders. • Gather background information based on the possible impacts and potential issues outlined in table 7.2. • Additional consultation survey work may need to be undertaken. • Confirm possible impacts and potential issues identified in SIA are relevant to the particular location. • Engage with key stakeholder to understand likely impact on their key areas of interest and accommodate, where possible, their needs and aspirations.
Step 3: Identify the assessment needed	• The detailed assessment of the chosen site will be informed by the findings from the ecological and economic research, the involvement of relevant key stakeholders, and further survey work outlined. • Level of risk needs to be understood before relevant management plans can be developed.
Step 4: Identify actions Required	• Identify what actions need to result from the social impact assessment? These need to be to issues that can be mitigated now and potential impacts that will need to be monitored and reviewed over time.
Step 5: Develop management plans	• The level of risk identified through the SIA (low, medium or high) should directly relate to the level of detail of the management plan- the higher the risk the more detail required within the plan.
Step 6: Consultation	• Develop a consultation strategy to manage on going impacts relating to the impacts of alien species. • The strategy needs to: - Develop and implement a communication strategy to provide timely and accurate information; - Ensure that the key stakeholders are “kept in the loop,” given their interest and investment to date; - Make sure that information is reviewed and updated with stakeholders at various stages of the project to minimise potential impacts. - Make the information outlined above available at the relevant local council offices and relevant websites.
Step 7: Monitoring and Review	• Monitoring of each impact identified in this report is a key tool in the success of this report.

7. Economic impacts

7.1 Introduction

This section reviews the economics of the alien fish species redfin perch, gambusia, tench, roach, streaked goby and yellowfin goby in Australia. The economic analysis relied on available data and existing international economic studies available at the time of the review.

Economic analysis provides a framework to analyse the impacts of alien fish species on the economy and local communities. It measures overall 'wellbeing' impacts and uses monetary or qualitative values to rank people's preferences for different management options.

In theory, economic analysis provides information to feed into:

- rational decision-making over a range of public intervention strategies;
- assessment of market and non-market effects of a particular alien fish species;
- quantitative and/or qualitative valuation of non-market impacts;
- choice of most efficient allocation of resources given socio-economic constraints;
- achieving the objectives of the decision-maker; and
- evaluating the effectiveness of research, control and prevention management programs.

Economic analysis identifies whether it is worthwhile managing a particular alien fish species because of its impacts and how best to do it. It also provides a common framework with which decision-makers can evaluate environmental, social and economic effects, in order to make a decision.

Economic analysis highlights the distributional effects of costs and benefits, i.e., the 'losers and winners' of each management option. This helps funding decisions.

A thorough economic analysis is conditional on a clear definition of the causes and consequences of alien fish species and the economic forces behind them (Horan et al. 2002). Biological and ecological data analysis, and determining the decision-maker's objectives in terms of management strategies, needs to take precedence over the economic analysis.

The extent of analysis of the alien fish species is dependent on data, timeframes and resources available. It is also dependent on when and where the alien species creates an impact as such species may have values in some locations but become a pest in others. The economic analysis on the alien fish species reviewed in this report has been limited by the quality and quantity of data availability.

7.2 Economic impact assessment methods

Economic impact assessments are a type of economic analysis. Supporting these assessments are economic valuation studies using revealed and stated preference methods. The former compare options on the merit of triple bottom line impacts, whereas the latter quantify impacts to feed into the assessments.

These methods and processes are used when comprehensively describing advantages and disadvantages (benefits and costs) from different actions, options or projects. Economic assessment methods compare the economic, environmental and social impacts of different management options either qualitatively or/and quantitatively over time. Common methods include:

- cost benefit analysis (CBA)
- cost-effectiveness analysis (CEA)
- multi-criteria analysis (MCA)

Both CBA and CEA work best with monetary values, whereas MCA does not require the assignment of monetary values to impacts. What the methods have in common is that all impacts are considered, quantitatively and/or qualitatively.

CBA estimates costs and benefits for management options over time and assigns present values to them. Typically a ‘do-nothing’ scenario is compared to a range of pest fish management options available to a regulatory body. CBA provides a transparent and objective framework to compare projects on common economic criteria (Mumford et al. 2000).

CBA is used mainly by governments and international agencies to determine whether or not particular projects or policies improve society’s welfare. CBA provides information to feed into decision-making, but does not represent the decision making process *per se*. In New Zealand the Biosecurity Act 1993 recommends the use of CBA to inform quarantine decisions. In Europe, the European Commission recommends CBA as an approach to decisions on biodiversity damage (EC 2000).

CEA is used either to determine the maximum benefits that can be obtained from a specified expenditure, or the minimum expenditure required to achieve a chosen outcome. For example, CEA could be used to maximise the impact of control for a given expenditure for a given alien fish species. Alternatively, it could be used to determine the minimum cost required to achieve a certain level of control (Corfield et al. 2008).

MCA evaluates management options in terms of the impacts of alien fish species using a set of sustainable development criteria. Each option is assessed qualitatively against scientific data combined with expert opinion. The results are then weighted and ranked by experts and/or the community to reach consensus following the well-structured process of the MCA.

MCA can provide a simplified decision making framework for the community to consider, assess, and weigh more complex issues such as alien fish species management. The flexibility of the tool allows for criteria to be modified and accommodates preferences through a weighting system.

Discounting: When undertaking a CBA, the choice of discount rate becomes paramount. The cost of using resources to control alien fish species means that these funds cannot be used for another activity. All costs therefore are opportunity costs (Kerr and Sharp 2006). Discounting adjusts opportunity costs: the costs and benefits of a given policy need to be discounted. A dollar benefit or cost in 20 years is not worth the same in today's dollar value.

The management of alien fish species has similarities with reducing greenhouse gas emissions to combat global warming. Both attempt to prevent long-term environmental problems, and there is uncertainty around predicting impacts (Keller et al. 2007). High discount rates and a lack of information on biodiversity values tend to undervalue benefits of alien fish species management occurring later in time (Turpie 2004). This tends to favour lower discount rates.

The choice of a discount rate remains the subject of extensive debate in economics (Boardman et al. 1996). Official guidance on a social discount rate is given by the United Kingdom's Treasury: 6% in real terms (HM Treasury 1997). However, for dealing with long term environmental issues such as pest fish much lower social discount rates (e.g., 2.5% to 3%) have been proposed (Pearce and Ulph 1999).

Latest research shows that economists favour discount rates declining over time, known as hyperbolic discounting (Keller et al. 2007). Given long timeframes, either hyperbolic discounting or a range of constant lower discount rates is recommended for assessing pest fish management options (3%, 6% and 9%). Sensitivity analysis allows for variations in cost benefits over time given different discount rates.

The conventional discounting formula is:

Present value = (Future value at time t)/(1+r)t where r is the discount rate and t is the time in years.

Inter-generational equity: There is generally a delay between the introduction of a fish species and its recognition as a pest. If a 'no control' strategy benefits the current generation (as it does not impose costs), but imposes large costs on future generations (loss in biodiversity), then compensation of the later generation becomes difficult.

The concept of inter-generational equity can be represented by viewing solutions to pest fish with a timeframe of at least 10 to 100 years. Pest fish are rarely eradicated and damages are borne for long periods. Benefits may be spread throughout the public over many years, while the costs of control are more immediate. This needs to be reflected in any economic analysis of pest fish impacts.

Longer timeframes are realistic, when looking *ex-post* at the eradication of the Coypu, which was introduced to the United Kingdom in the 1920s for its fur. Its population peaked at 200,000 in the 1950s and it was successfully eradicated by 1987 after nearly 70 years of invasion (Ciruna et al. 2004).

Distribution of costs and benefits: Identifying costs and benefits may prove a useful exercise for deciding on the level of intervention. However, value judgements and distributional questions of who pays for the intervention can delay decisions. The magnitude of costs may make intervention politically unacceptable, even when the benefits are likely to be even greater (Kerr and Sharp 2006).

Impact categories: Pest fish impact on ecosystems and sometimes local economies. They can adversely impact on industries, such as fisheries, tourism, aquaculture, and water quality. Pest fish can also contribute positively to recreational fishing and the enjoyment of fishing.

Often the full range of economic costs of pest fish goes beyond the immediate impacts on established industries such as fishing. There are secondary and tertiary effects of invasions: shifts in consumer demands, loss of biodiversity, decline in natural resource and environmental amenities (McNeely 2000).

Pest fish may affect markets (e.g., changes in prices), and non-market attributes (e.g., changes in ecosystem services). Market impacts can result in production losses (e.g., decreases in fisheries and aquaculture production, native fish, water quality and property values). Non-market impacts include potential declines in natural capital resulting from losing ecosystem services (Ciruna et al. 2004).

When pest fish affect industries, costs are more straight-forward to estimate. The value of lost production is market priced. Comprehensive valuation exercises will encompass a range of both market valued (e.g., lost production) and non-market valued (e.g., decline in native fish) goods and services (Horan et al. 2002). However, estimating the value of non-market impacts can be difficult, time consuming and costly (Ciruna et al. 2004; Evans 2003; McNeely 2000). Nonetheless, all the potential impacts of pest fish should be considered, at least qualitatively (Ciruna et al. 2004).

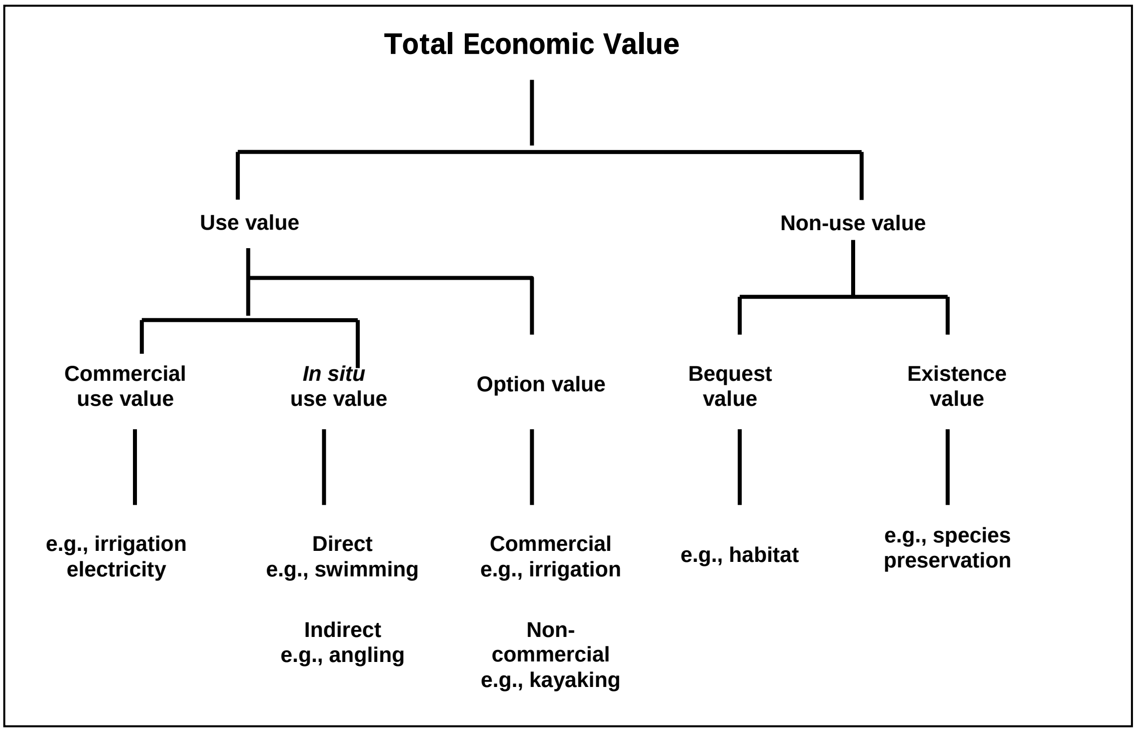
7.3 Review of economic valuation methodologies

Economic valuation methodologies: Economic valuation attempts to put monetary values against pest fish impacts to be used in economic assessments. Studies typically address three fundamental questions before choosing the most appropriate valuation methodology: what should be valued, when, and to what extent (Kerr and Sharp 2006).

Economic valuation studies only measure a given change in value, rather than absolute values. For example, there is no 'value of biodiversity' as such, and most studies measure the relative change in the value of biodiversity as the result of a pest.

There are numerous valuation studies but studies covering the total effects of pest fish on ecosystems and economies are nearly non-existent. Also, quantifying damage to ecosystems is still an emerging science. A more controversial study has been the valuation of the world's ecosystem services at US$36 trillion (Costanza 1997).

Total economic value approaches: Valuing ecosystems has been a useful political and awareness raising tool, but the environmental and resource economics concept of the Total Economic Value (TEV) is a more suitable analytical framework (Turpie 2004). Rather than providing a lump sum valuation, it assigns monetary values to different components. TEV breaks down the value of a given species into use values (commercial, 'in situ' and option values) and non-use values (bequest and existence values). Figure 7.1 shows the TEV framework including use and non-use values of improved water quality.



Source: Adapted from Ministry for the Environment (2005).

Figure 7.1: Total economic value framework of improved water quality.

Daily (1997) provides an example of 'use' and 'option' values (e.g., the potential value of such services in the future) of ecosystem services. Alien fish may affect locally important ecological services by affecting the hydrological cycle, including flood control and water supply, waste assimilation, recycling of nutrients, conservation, and regeneration of soils (Daily 1997).

There are two types of approaches for valuing use and non-use values: revealed preference and stated preference approaches (Kerr and Sharp 2006).

Revealed preference approaches: Revealed preference approaches select the best possible option on the basis of consumer behaviour (Varian 2005). Preferences of consumers are revealed by their purchasing habits (e.g., fishers reveal their preference for fishing by buying fishing gear). Therefore only things for which there is an associated market can be valued. Fishers invest in time and travel services to go fishing and these costs are used to derive recreational values (e.g., via Travel Cost Methods) (Kerr and Sharp 2006).

The impact of pest fish on home prices can potentially be used to measure the benefits of a control program (Hedonic Price Methods) (Kerr and Sharp 2006). Revealed preference approaches depend on data collection methods and are subject to time and budget constraints.

Stated preference approaches: These rely on responses in hypothetical games (e.g., fishers are asked how much they are willing to pay for a given increase/decrease in quantity of a particular pest fish in a lake). Methods include contingent valuation, choice modelling, contingent ranking and rating, and conjoint methods (Kerr and Sharp 2006). Choice modelling (CM) is more versatile in its application than its counterparts.

CM allows for an estimation of a wide range of values - including use and non-use values - thus making it more cost-effective. CM studies have estimated non-market and social values associated with environmental management strategies. CM would be preferable for pest fish valuation.

Stated preference methods measure values without existing markets and hypothetical outcomes of control strategies. Similar to the revealed preference methods, stated preference methods depend on data availability and are subject to time and budget constraints.

Benefits transfer approaches: Benefits transfer (BT) uses a more rapid but less accurate valuation process. BT extrapolates results or data from the context of one or several existing valuation studies (defined in terms of their time frame, location, environmental quality change, and/or affected species) and transfers them to a context that is specifically relevant for a pest fish policy (Kerr and Sharp 2006).

BT has two main potential advantages in that it is faster and less expensive than stated and revealed preference approaches. However, difficulties are likely to arise (e.g., lack of high quality studies to draw on, differences in context, transfer biases).

Given that alien species valuation studies are sparse and those specific to pest fish even more so, the use of BT will be challenging. It will depend on the intended [use of the benefit?] whether this method is appropriate. Some situations benefit from an indication of costs rather than precision policy application.

Figure 7.2 summarises the implications of the economic framework valuation for pest fish.

- Valuation measures marginal changes in quality/quantity, not absolute values.
- Policy objectives need to be clearly defined before any valuation can take place.
- Impacts of pest fish on biodiversity are likely to be wide-ranging; therefore any valuation framework should consider pest fish within the context of a broad ecosystem (not a small one where biodiversity is lower).
- The type of people-preferences surveyed will affect the valuation.
- The impacts on ecosystems over time, and lag effects of pest fish need consideration.
- While first round effects are important, the valuation exercise should at least consider second round effects.
- Revealed and stated preferences methods are resource intensive but deliver more reliable data.
- Benefit transfer is faster and cost-effective, but relies heavily on high-quality

Figure 7.2: Implications for the economic valuation of pest fish.

7.4 International literature review of economic impacts studies

There is an abundant international literature that measures changes in value resulting from invasions. The key values estimated include recreational use values, ecosystem function values, and existence values for flora and fauna. This shows that people do place economic values on environmental changes attributable to biological invasions (Kerr and Sharp 2006).

There are few attempts to aggregate the TEV of aquatic invasions. There are even fewer studies valuing the economic and social impacts of pest fish in Australia. Table 7.1 provides examples of economic impact estimates of selected invasive species.

Table 7.1 Examples of economic cost estimates of invasive species including freshwater fish.

Species and Location	Economic Variable	Economic Impact US$ (unless stated)	Reference
(A) General			
• 79 harmful invasives including sea lamprey, zebra mussel, Asaina clam, purple loostrife, melalluca, and hydrilla • 3 harmful fish • 3 aquatic invertebrates • aquatic plants	• Total cumulative damages	• $131–185 billion • $631 million • $1.6b • $135m pa	OTA 1993
• All invasive species in U.S. • 138 Fish	• Economic damages and control costs(used 10-times the number of species than OTA)	• $120b pa • $5.4b pa	Pimentel et al. 2005
• Wide range of species including mammals, birds, amphibians, fish, molluscs, plants. • Aquatic weeds • Fish	• U.S. total damage • Losses and damages • Losses and damages	• $147b pa • $10m pa • $1b pa	Pimentel et al. 2000
• Pest animal vertebrates in Australia	• Includes some environmental costs, but excludes social impacts e.g., human health costs	• $0.72b	Agtrans Research 2005
• 20 invasive species in Germany	• Annual direct economic damage and control costs	• €167m	Reinhardt et al. 2003
• Six weed species	• Costs in Australian agroecosystems	• $105m pa	Watkinson et al. 2000
(B) Aquatic Species			
• Zebra mussels	• Damages to US and European industrial plants 1988 – 2000	• $0.75 to $1b	O'Niell 2000
• Zebra mussel	• Research expenditures	• $10.9m pa 1992-4	Hushak and Yuming 1997
• Zebra mussel	• U.S. Fish and Wildlife estimate	• $5.5b over 10 years 1990-0	Sun 1994
• Ruffe	• Estimated loss to sport fishery in Lake Erie • 1985, 1995	• $724m cumulative	Hushak 1997
• Ruffe	• Losses for native fisheries in Great Lakes	• $520,000 pa	Jenkins 2001
• Carp	• Total impact costs • Environmental costs	• $16m • $11.8m	Agtrans Research 2005
• Sea lamprey	• Lost fishing opportunities and indirect economic impact of terminating control	• $675m pa	OTA 1993
• Sea lamprey , Great Lakes fisheries, U.S. and Canada	• Loss of fishing opportunities and indirect economic impacts	• $500m pa	Spaulding and McPhee 1989
• Sea lamprey, Great Lakes, U.S. and Canada	• Not specified	• $13.5m pa	Jenkins 2001
• Sea lamprey New York and Michigan	• Not specified	• $304,000 • $3.3m pa	GAO 2000
• Sea lamprey St. Mary's river	• Total costs in 2015	• $3.2-$5.8m	Lupi et al. 2003

Species and Location	Economic Variable	Economic Impact US$ (unless stated)	Reference
• Comb-jellyfish	• Damaged the anchovy fisheries in the Black Sea	• $17m pa	Knowler and Barbier 2000
• Coypu aquatic rodent	• Impacts on agriculture and river banks in Italy	• $2.8m pa	Panzacchi et al. 2004
• Bugula Neritina in California	• Per pound of finished product (pharmaceutical value)	• US$376,390 (benefit)	Marsa 2002
• Aquatic plants in Florida	• Per acre welfare change due to reduction in commercial red king crab	• US$6,345	Thunberg and Pearson 1993
(C) Aquatic Weeds			
• Water hyacinth	• Costs in 7 African countries	• $20-50m pa	Joffe-Cook, 1997
• U.S. aquatic weeds	• National impacts	• $1b -$10b	Rockwell 2003

Cost estimates shown in Table 7.1 are not negligible – pest fish are costing economies millions, possibly billions of dollars per annum. For example, Pimentel et al. (2000) estimate that the total damage costs to the US of invasive species is $147 billion per year, of which invasive fish account for $1 billion annual losses and damages.

The extent of pest fish and their impacts globally is considerable. When adding control costs to economic damage, 138 fish species introduced to the United States cost the economy around $5.4 billion annually (Pimentel et al. 2005). As a result, forty-four native species of fish are threatened or endangered by pest fish (Pimentel et al. 2000).

In the United States, the introduction of ruffe to Lake Erie caused cumulative losses of $724 million to sport fishery alone (Hushak 1997). The same fish is responsible for $520,000 per annum losses to native fisheries in the Great Lakes (Jenkins 2001).

A number of research papers show that invasive species are costing Australia alone ‘many millions of dollars annually’, mainly in costs of control and value of production foregone (Agtrans Research 2005). The only economic impact estimates found for introduced freshwater aquatic vertebrates were for carp: $11.8 million per year of environmental costs were derived by aggregating estimated costs of carp-related sedimentation and heightened water turbidity. Total impact costs are estimated at $16 million per annum (Agtrans Research 2005).

Another cost component is the expenditure on research to identify control methods and management requirements. For example, the majority of the costs of managing gambusia lie in gaining a better understanding of their effects and dispersal patterns, rather than direct control (NPWS 2003). Research costs often extend over a number of years (e.g., research on gambusia control costs on average $22,000 a year, with studies likely to be longer than five years) and need to be factored in total costs.

A point to consider is that all costs estimates vary widely. Variations in costs could be explained by the:

- extent of analysis (type of costs included: control, recreational loss, direct/indirect and environmental/economic);
- type of species surveyed (terrestrial versus aquatic, aquatic weed versus fish);
- location (at country or ecosystem level);
- spatial dimension of costs (annual versus cumulative); and
- year of survey and methods used.

Inconsistencies can be explained further by the fact that the aggregate cost of invasions are made up of many variables, which are subject to errors of measurement (Horan et al. 2002). Adding different cost categories (market and non-market) and type of measurements (qualitative and quantitative) is not always a straight-forward exercise.

Most studies have concentrated on terrestrial or riparian invaders, rather than aquatic species (Turpie 2004). The current lack in alien fish valuation studies jeopardises transferring sensible values through benefits transfer. This will make the use of benefits transfer less favourable; future economic valuations need to lean towards more expensive and time consuming valuation exercises such as revealed and stated preference studies, before undertaking CBA or CEA.

The majority of estimates of the different costs seem to be incomplete and existing ones need refinement and further justification if used in policy making. Often environmental or social costs are excluded (Agtrans Research 2005). The majority of cost estimates do not account for lost biodiversity values (Normile 2004). This suggests the need for further economic valuations studies to be commissioned.

Sometimes some costs have been neglected. For example, the loss of genetic information is not often considered (Perrings et al. 2002). There are few estimates of the magnitude of these costs even though most studies indicate that these economic costs are not negligible (Perrings et al. 2000). Nonetheless their valuation is challenging, and they need to be considered at least qualitatively for policy formulation.

Some costs are harder to measure: the recreational value of fishing can be derived from fisher behaviour and consumption patterns, but the social value of the fishing experience itself is often excluded. It can be argued that some experiences are priceless and valuation attempt would fail to represent the 'true' value.

Although some alien fish species have reduced the numbers of some native fish species and driven others to extinction, some of these fish do provide marginal economic benefits for sport fishing.[7] At the same time more than 40 non-indigenous species negatively affect native fishes and other aquatic biota in the US and create a loss to native sport fishing (Pimentel et al. 2000). Economic losses due to pest fish in the U.S. are estimated at more than $1 billion annually (see Table 7.1).

Even though very few studies estimate the full range of costs and benefits of pest fish, the potential costs implications are still considerable Table 7.1) and in most cases outweigh the benefits of having the species. This could apply to the redfin perch and gambusia. While cost estimates might appear considerable and warrant action, it is still important to consider both costs and benefits with policy options and relative values (Kerr and Sharp 2006). Given the level of uncertainty about the severity of pest fish, decision makers, economists and scientists will face difficulties in choosing the optimal management strategies (Knowler and Barbier 2000).

7.5 Economic impacts of alien fish species

Our review of the ecological impacts of tench and roach in Australia indicated that these species need not be considered as pest species at present, at least given their current restricted distribution. Similarly, too little is known about the streaked and yellowfin gobies to classify them as pest fish species at present. However, it is apparent that both redfin perch and gambusia can have significant impacts on the indigenous fauna in some habitats and in this respect can be regarded as pest fish species. Knowledge of the economic impacts and cost/benefits of these two species is therefore required.

7.5.1 Economic impacts of gambusia

Gambusia were introduced in the 1920s for mosquito control, but are now widely regarded as being relatively ineffective for this purpose in most locations; they represent a significant pest in freshwater rivers and streams (chapter 4). Gambusia have a reputation for eating mosquito larvae and for this reason were introduced globally. Recent literature has shown that they negatively affect native fish species and are no more effective at mosquito control than native fish species (see chapter 4).

This indicates that the introduction of gambusia has not generated additional economic benefits, which other native fish were not already able to deliver. Therefore the

[7] In the U.S. sport fishing of all fish species contributes $38 billion annually to the economy (Pimentel et al. 2000).

benefits of mosquito control by gambusia are likely to be negligible or at least comparable to other native species. Contrary to redfin perch, gambusia do not generate any value for recreational fishers and therefore recreational benefits are likely to be negligible.

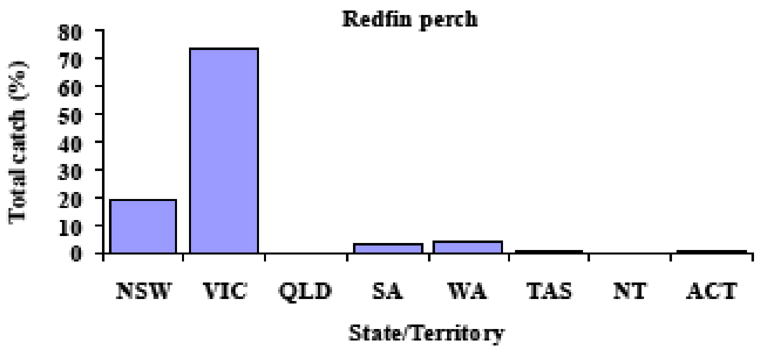
In contrast, gambusia is clearly responsible for a reduction in indigenous fauna including some rare and threatened fish and amphibian species (see chapter 4). One of the few positive effects is that ‘preferential predation by redfin perch on gambusia appeared to provide competitive benefits to native species such as carp gudgeons (Smith and Hammer 2006).

7.5.2 Economic impacts of redfin perch

No economic valuation studies on redfin perch were found, but some information on recreational fishing is available and so estimates of recreational benefit can be derived from these.

Annual harvest and total catch: Between May 2000 and April 2001, Australian fishers harvested a total of 136 million of aquatic animals including finfish, small baitfish, crabs and lobsters, prawns and yabbies, cephalopods, molluscs and other species (Henry and Lyle 2003). This included 60.4 million finfish, of which the total catch for redfin perch amounted to 2.26 million or 3.74% (Henry and Lyle 2003).

The annual harvest for redfin perch from recreational fishers was estimated at 1.3 million fish (Henry and Lyle 2003). Approximately 73% were caught in Victoria (VIC), 19% in New South Wales (NSW), whereas SA, WA and TAS averaged 1-4% (see Figure 7.3). Total catch numbers are made of the annual harvest plus any released and/or discarded fish. Table 7.2 shows annual harvest numbers and total catch for redfin perch for selected Australian states by recreational fishers.



Source: adapted from Henry and Lyle (2003)

Figure 7.3: Relative distribution of the annual recreational harvest (based on numbers) by state and territory fished for redfin perch (Henry and Lyle 2003).

Annual catch of redfin perch totalled 2.26 million including an annual harvest of 1.3 million. VIC had the highest annual total catch at 1.66 million followed by NSW at almost half a million.

Table 7.2: Estimated annual harvest numbers and total catch (including released/discarded fish) of redfin perch by Australian recreational fishers.

State	VIC	NSW	WA	SA	TAS	ACT	TOTAL
Annual harvest	949,351	244,596	47,384	40,410	9,316	3,454	1,294,511
Total catch[1]	1,660,267	427,475	82,812	70,624	16,281	6,036	2,263,496

[1] Total catch calculated from data from The National Recreational and Indigenous Fishing Survey including released/discarded fish.

Expenditure: Surveys on the expenditure incurred by recreational fishers are used to assess the economic impact of redfin perch on recreational fishing and the flow-on benefits to the local economy. The 2003 National Recreational and Indigenous Fishing Survey estimated total annual expenditure on services and items for recreational fishing at $1.8 billion (Henry and Lyle 2003). The national average attributable expenditure was estimated at $552 per fisher per annum.

The largest individual expense for fishers was boats and trailers ($872 million), followed by travel ($432 million), accommodation ($184 million) and fishing tackle ($146 million) (Henry and Lyle 2003).

Table 7.3 shows the number of fishers, total attributable expenditure, redfin perch catch rates and average annual fisher spending for selected states. Based on these figures, the proportional total catch and attributable expenditure to redfin perch for each state has been calculated.

Table 7.3: Attributable recreational fishing expenditure for redfin perch for selected Australian states.

State	Total fishers (N)	Total redfin perch catch (N)	Redfin catch (% of total catch)	Total attributable expenditure ($M)	Average fisher spending per annum ($)	Redfin perch attributable expenditure $
NSW	998,501	427,475	0.314	554.204	555	1,741,860
VIC	549,803	1,660,267	1.221	396.27	721	4,839,287
WA	479,425	82,812	0.061	338.38	706	206,101
SA	328,227	70,624	0.052	148.48	452	77,042
TAS	124,590	16,281	0.012	51.83	416	6,205
ACT	53,467	6,037	0.004	19.36	362	859
TOTAL	3,319,058	2,263,496	1.664	1,508.52	552	6,871,354

Redfin percentage catches were calculated as a percentage from the total catch of 136 million. For example, for Victoria the catch of 1,660,267 redfin perch represented 1.2% of the total catch of 136 million. Expenditure attributable to redfin perch is then derived from:

Fisher number x average fisher spending per annum x redfin catch % of total catch[8]

Estimated total expenditure for recreational redfin fishing amounts to $6.87 million per annum for the selected states. State expenditure is related to the average annual fisher spending and number of fish caught. VIC has the largest redfin fish expenditure ($4.84 million) and Australian Capital Territory (ACT) the smallest ($859)[9].

Expenditure includes fixed and variable costs (i.e., the purchase of bait, fuel, travel, accommodation and fishing gear, boats and annual licences and fees). Redfin perch are predominantly caught on lines. Over 85% originate from lakes and dams and the rest from rivers (Henry and Lyle 2003). The catches are equally divided between on-shore and offshore (e.g., by boat) locations.

The expenditure data do not provide a full estimate of the value for redfin perch as they exclude the enjoyment fishers derive in addition to fish catching; therefore they are likely to represent an underestimate of the total value of the recreational fishing experience. The estimated recreational benefits of $6.87 million indicate the value

[8] Redfin perch attributable expenditure can also be calculated as:
total attributable expenditure x redfin catch % of total catch

[9] This is based on the assumption that fishermen spent equal dollars for fishing effort regardless of the species caught.

fishers attach to catching redfin perch. This is only a derived estimate; a direct survey of the recreational value of redfin perch would provide more accurate values.

In the National Recreational and Indigenous Fishing Survey, recreational fishers listed several other motivations such as 'to relax and unwind', 'fishing for sport', 'to be with family' and 'to be outdoors' (Henry and Lyle 2003). Economic valuation surveys would attach additional $ benefits to these experiences. Only a few fishers considered catching fish for food as their primary motivation (Henry and Lyle 2003).

Scientific observations have shown that redfin perch are predators of native fish species and can affect ecosystems to various degrees (chapter 4). In addition, perch can compromise the development and viability of sports fisheries based on stocked trout (Molony et al. 2004). No monetary estimates of these impacts are available. Based on existing studies of other pest fish impacts, these negative non-market values are likely to be considerable. Further economic and scientific investigations will have to be undertaken to confirm this, before the estimated recreational benefits are used for decision-making.

7.6 Management control options and costs

There are a number of management options for mitigating, managing or controlling the effects of pest fish (see chapter 8). Examples of control options taken from the Australian experience of researching carp management choices (Harris 1997) include (in decreasing scale of intervention):

Eradication: the pest fish is completely removed from an area by a time-limited campaign. The coypu in the UK has been eradicated in this way (Panzacchi et al. 2004).

Strategic, site-specific control: releases a biocontrol agent that controls in perpetuity. This can be a cost-effective solution if the organism can be effectively controlled within a given timeframe and the cost of management is less than a prescribed amount. In New Zealand, the use of rotenone to eradicate koi carp and gambusia in the Nelson area (and hence in the South Island), and alligator weed infestations in the Bay of Plenty, are examples of this strategy (Harris and Skilton 2007).

Strategic sustained control: reduces the pest fish to low numbers and sustains them at the specified number. This can include ongoing netting or harvesting programmes.

Strategic, targeted control: implements control when conditions are desirable (e.g., numbers are about to increase or damage is about to occur).

Crisis management: control is only done reactively, lacking proactive planning and typically incurring high costs.

Do nothing: the costs of control exceed benefits and no intervention may be the best option.

The level of intervention will depend on risk assessment and perception; and the resources available to the relevant agency.

Generally the costs of control are difficult to define due to data scarcity, lack of monitoring and record keeping and little *ex-post* analysis. Often control costs are site specific and vary according to the type of species and area controlled. Trends in costs over time are rarely monitored. Analyses *ex-ante* and *ex-post* are indicators of control costs and economic impacts over time. It is important to monitor and aggregate the monetary costs of pest fish over time.

There are few studies looking at the cost estimates of control for pest fish, even less so for gambusia, redfin perch, tench, roach or the gobies. One example is the 2003 NSW threat abatement plan for *Gambusia holbrooki* (NPWS 2003). Control costs were estimated to be AU$220,000 over five years, averaging $44,000 a year. Half the costs were attributed to research, the remaining being split between monitoring and control costs such as creating supplementary habitat and chemical control trials.

The majority of costs were to be spent on gaining a better understanding of the effects and dispersal patterns of gambusia, rather than on direct control (NPWS 2003). However, because research on and actual control of gambusia is likely to go beyond five years, total control costs can be expected to be much higher.

Table 7.4 shows the estimated costs and benefits of targeted invasive species control from a number of studies. The control costs of ruffe could serve as an example of the potential control costs for redfin perch. In the US, annual costs of controlling ruffe are around US$1.2 million per annum over an 11 year program. The socio-economic situation in Australia is different to the US, but these costs still serve as an indication of their potential magnitude.

Increased investments in managing invasive species, particularly aquatic species, can be easier to justify in economic terms than the benefits of non-control (McNeely 2000; Turpie 2004). It is the funding of actual control programs that is harder to justify and finance. Investments will have to be paid for by the community, who may not always directly benefit.

Table 7.4: Control costs and benefits of different methods from selected studies.

Targeted Pest Species	Control Method	Estimated Control Costs	Reference
(A) General			
U.S. non-indigenous pests	Pesticide applications	$7.4 billion pa	OTA 1993
16 invasive species in Canada	Cost of control and eradication	$13.3 and $34.5b pa	MacIsaac 2004
(B) Aquatic vertebrates			
Gambusia	5 year control costs (including research and monitoring costs)	AU$220,000	NSW 2003
Ruffe	Benefits of control programs to sport and commercial fishing	US$119m to US$1.05b cumulative (benefits)	Leigh 1998
Ruffe	Estimated total cost of 11-year program	US$13.6m cumulative	Leigh 1998
Carp	Management and research costs	$4m	Agtrans Research 2005
Brown trout, Little Kern River	Eradication to protect golden trout population	US$1m since 1985	OTA 1993
Sea lamprey, Great Lakes	General control programs	US$2.1-$4.3b pa (benefits)	Sturtevant and Cangelosi 2000
Sea lamprey	Control and research to reduce predation fish stocking	US$10m pa	OTA 1993
Sea lamprey, St. Mary's River	Sterile male release and trapping	$300,000 pa	Lupi. Hoehn, and Christie 2003
(C) Aquatic invertebrates and weeds			
Aquatic weeds, majority of which are non-indigenous in U.S.	General control	$100m pa	OTA 1993; Pimentel 2000
Zebra mussel	Monitoring and control costs for 339 industrial facilities	US$83m and a mean of $248,000 per facility	O'Neill 1997
Zebra mussel, Great lakes, tributaries and inland waters	125 facilities	US$509,000 1998-94	Hushak et al. (1995)

Approaches to estimate control costs have not always been consistent. Multi-species studies (OTA 1993; Pimentel et al. 2000; MacIsaac 2004) have aimed at highlighting the damage done by invasions or demonstrating the value of removal. Single-species studies have generally analysed the costs and benefits of biological control programmes (see Table 7.4). The latter have been presented in terms of cost–benefit ratios, while the former have also been given in terms of annual values before and after invasion and net present values with or without control programmes, usually using CBA (Turpie 2004).

The majority of control costs are large, but highly variable. In his review, McNeely (2000) found that the cost-benefit ratios of control methods seem to vary widely. This could be explained by the use of different control methods, the country of control, the assessment method and the area covered. In nearly all cases the benefits of control far outweigh the costs involved, justifying intervention (McNeely 2000). Estimates of control costs need to be undertaken for each specific site or region when applying them to the pest fish.

7.7 Gap analysis

This economic analysis is only a preliminary assessment of the pest fish, but has drawn attention to a number of gaps in existing economic impact analyses of pest fish and decision making processes. Some of these gaps are highlighted:

Lack of pest fish economic impact studies: Currently there are insufficient studies to determine the economic impacts of pest fish at either regional or national level (Lovell and Fernandez 2006). There are very few studies done in Australia, and even fewer on the pest fish in question.

Dearth of pest fish economic valuation studies: The ‘willingness to pay’ method of valuation is used only sparingly by planners and policy makers. Most, if not all, studies have targeted terrestrial or riparian invaders, rather than aquatic species. No specific valuation studies for gambusia, redfin perch, gobies, tench and roach are available.

Full cost analysis needed for redfin perch and gambusia: The recreational value of $6.87 million gives an indication of some of the economic benefits of redfin perch. This excludes other values such as the enjoyment of fishing, but more importantly, it does not include the negative impacts associated with predation on native fish species and impacts on ecosystems.

The same is true for gambusia, the only difference being that gambusia’s recreational benefits are negligible. No monetary estimates of environmental impacts have been found. Based on existing studies of other pest fish impacts, these negative non-market values are likely to be considerable. Further economic investigations have to be undertaken to confirm this.

For the gobies, tench and roach, there is too little scientific data available to draw any conclusions on economic impacts. Unless further biological and ecological assessments are carried out, any conclusions will be premature.

Lack of resources: There seems to be a lack of general resourcing for pest fish prevention programmes, control strategies and research. In the US, implementing and enforcing state laws are seen as ‘less’ or ‘much less’ than adequate (OTA 1993).

7.8 Recommendations

There is a lack of valuation studies and economic impact studies for the alien pest fish species reviewed here. Most studies do not go beyond recreational benefits measures for non-market impacts. The more intangible ecosystem attributes, such as environmental or social impacts, values of indirect costs of control measures, and potential effects on industries other than primary industry (e.g., the tourism industry), are rarely included. Improved estimates are required in order to provide a reliable estimate of the costs and benefits of existing alien pest fish in Australia, and the potential economic impact of their introduction to new water bodies.

There is an urgent need to commission high-quality studies in order to support policy formulation. Valuation exercises like contingent valuation studies need to follow internationally accepted guidelines (Turpie 2004). Choice modelling seems to be a more flexible and cost-effective valuation method.

In terms of intervention policies, biologists often advocate for preventative rather than control measures because of the paucity of effective control methods and their high costs. Additional economic analysis is required to better understand different intervention strategies and control programs to allow for comparison. The true challenge may not lie in determining the precise costs of the impacts of alien pest fish, but in estimating prevention costs and benefits.

Focusing resources in determining the cost-benefit ratio of controlling the alien fish species may not prove as effective as deciding on the extent of management regimes. It may prove more productive to identify sites at risk of high ecological and biodiversity loss and optimise resource allocation accordingly, particularly when data are missing.

The ability of economists to provide useful analyses depends largely on how well scientists are able to estimate the probabilities of future impacts of alien pest fish species in a consistent and comparable way. Economic models provide little assistance when based on unclear predictions of biological events; an effective partnership between scientists and economists is essential.

7.9 Summary and conclusions

Economic analysis provides a framework for decision-makers to choose between a range of prevention, research and control strategies for the alien pest fish species. However, economic assessments of the redfin perch and gambusia need to be backed by sound ecological and biological information. The decision-makers' objectives need to be clearly laid out before detailed economic analysis can be undertaken.

Cost-benefit and cost-effectiveness analyses are most effective where data are available and of a high standard. Multi-criteria analysis compares intervention levels qualitatively and allows for stakeholder feedback. Extensive data collection is recommended for high priority sites.

For economic valuation, choice modelling is more versatile in its application than its counterparts and would be preferable for valuing impacts of individual alien fish. Benefits transfer uses a more rapid and cost-effective but less accurate valuation process. The lack of studies to draw from will make the use of benefits transfer challenging.

Gambusia: It is clear that gambusia is a pest species (chapter 4) and that it is desirable to prevent its further spread while eradicating it where possible. An assessment of economic benefits is not required for this species as there are few. However, the economic costs/benefits of various management options (e.g., do nothing, contain its spread, control numbers, eradicate localised populations etc.) need to be determined, to inform future decision-making.

More detailed analysis of control costs at local and regional levels is essential to prioritise intervention efforts. Costs of prevention (e.g., avoiding spread to non-contaminated areas) and further research on effective control methods need to be estimated as part of total management costs.

Redfin perch: It is clear that redfin perch have significant economic values in Australia but they are also responsible for impacts on indigenous fauna (see chapter 4). Initial estimates of recreational benefits amount to $6.87 million per annum, but no direct estimates of damage costs were possible. The potential for controversy is therefore high and solid economic data will be required to support future management.

Thorough analysis is likely to be required on a site specific basis and to indicate whether the introduction of perch to a new body of water will be of overall benefit or not. Currently, it is difficult to place a dollar value on potential ecological damage, but methods available to managers to consider *a priori* costs/benefits of an introduction and/or control could include such valuation techniques as choice modelling, impact assessments and cost-benefit analysis. Considerable high quality data need to be collected to feed into any of these economic analyses.

Tench and roach: There is little point in carrying out an economic cost/benefit analysis on these species at present as they are minor fisheries and too little is known about their potential to become a pest fish species in Australia. At present there is no evidence of impacts (see chapter 4); but if these species are spread to other more suitable water bodies, they may then become a pest in these locations. In this respect, the feasibility and costs of containment as against eradication may need to be determined to inform future management.

Yellowfin and streaked gobies: Little economic analysis can be done until knowledge of impacts on marine ecosystems is available.

8. Control methods

8.1 Introduction

Control of pest fish involves two major components. Firstly, further spread needs to be prevented through measures such as legislation, policing, public education and the construction of physical barriers, where appropriate. Such measures are designed to prevent future impacts. The second component includes measures to reduce actual impacts that are occurring and this section deals with these.

Eradication of alien fish is often desirable but rarely feasible, and it may not be an essential part of managing an alien 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 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 alien 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. 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. Furthermore, where two species of alien fish are present in an environment, the reduction of one species may increase the abundance of the other resulting in different impacts. For example, a reduction in rainbow trout stocking in a New Zealand lake to reduce predation pressure on dwarf galaxiids (*Galaxias gracilis*) in the pelagic zone, resulted in an increase in gambusia in the littoral zone and an increased mortality of the galaxiids when they entered the littoral to feed in autumn (Rowe 2003). Research is needed to identify such constraints on pest fish control programmes but also 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. Ongoing 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 alien pest fish (see chapter 3), but they require knowledge of fish densities. A major limitation of the adaptive management approach to alien 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 carefully compared with the costs involved in the pest fish management program through a bioeconomic study. This is because the cost of control may 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 trout from Johnson's Lagoon 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 would 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. Judgment is required to make this comparison: this requires a clear appraisal of the ecological impacts, plus the consequences doing nothing (which 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 timeframes 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 harbour pests and diseases. RSPCA Australia acknowledges that in certain circumstances it is necessary to reduce or eradicate populations of some introduced animals; however, it maintains that 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; a reduction in the risk of re-introduction will then depend on education of the next generation.

Control strategies for the alien 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 both in Australia and abroad, though few of the six established 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 learned that can be applied to alien 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 which methods to use for which species in which places. Experience has indicated that the effective method, or combination of methods, will vary greatly depending on site and species-specific factors. Thus, this chapter reviews the potential choices of method that can be 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.

8.2 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. To be effective, these methods often require intensive effort, 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 used where other more effective methods of control are not practical or not supported. 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, compromising the potential for long-term control. 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 alien fish 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 alien 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 organisms. Gill netting only targets active fish and is not particularly effective on sedentary species or when fish are not actively moving. 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 (pers. comm. G. Closs).

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). Seine netting is typically used to sample fish in open waters and is particularly effective at capturing schooled or clumped fish such as gambusia (Anthony Moore, pers. obs.).

Tench are generally harvested using trammel nets, but seine and gill nets are also used. Balik and Cubik (2000) found that trammel nets with an outer wall of multifilament and an inner wall of monofilament were more efficient than other combinations. The catch per unit effort (CPUE) for trammel nets with monofilament inner walls of 28, 40, 50 and 60 mm mesh size was twice as high as for nets with an inner wall of multifilament netting (Balik 2001). Balik and Cubic (2001) also investigated gill nets. They found that red, yellow, brown and blue gill nets of nylon monofilament were more effective on tench than black, white, light green, or dark green nets.

The mean CPUE of tench in Lake Parkinson, New Zealand, caught by both fyke nets and Wisconsin traps, declined between 1976 and 1978 following quarterly netting, as did the mean size of the tench caught. Taken together, these results indicate a reduction in the number of fish over time, especially the larger-sized fish. Shag predation on tench in the experimental or treatment arm of this lake, which was stocked with grass carp, increased as weed cover was removed by the carp (Rowe and Champion 1994); but as a reduction in tench CPUE also occurred in the control arm of the lake where macrophytes remained, this is unlikely to have contributed much to the decline in tench. The decline in tench numbers between 1976 and 1978 was therefore attributed mainly to netting and trapping.

Trapping can generally be split into two primary types, namely: 1) small mesh traps to sample small bodied species, and 2) large traps to capture large bodied fish by the use of attractants. Each technique has its own advantages, though they tend to be dependent on both size of target species and location/habitat type. Small mesh traps tend to be used in situ whereas the larger traps are often used around the world to capture fish undertaking migrations to or from spawning habitats. Traps have been recently devised to catch migrant common carp in streams by forcing them to jump over an artificial barrier into a holding pen (Stuart et al. 2003). Netting was successful in reducing carp abundance in Lakes Cresent and Sorrel in Tasmania, but eradication is proving more difficult; whereas it may be possible in Lake Cresent, it may not succeed in the much larger Lake Sorrel (ASFB 2005). Fencing is now being used in conjunction with traps to prevent carp spawning and to enhance carp capture in traps in these lakes (Diggles et al. 2004). Radio tracking studies have revealed that most carp migrate through a narrow isthmus on one side of Lake Sorrell to reach spawning grounds on the other side and this presents an ideal opportunity for trapping (ASFB 2005). Trapping efficiency can be improved during spawning when used with attractants such as gravid females or pheromones taken from gravid females; this is happening in Lake Sorrell.

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. 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 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. Redfin, roach and tench were all introduced into Australia for their recreational angling value at the time; small-bodied species like gobies and gambusia are too small to be targeted via such methods. Therefore, line-fishing is a technique that probably has only a limited application for removal of established alien fish in Australia. With the public undertaking line fishing of a designated 'alien fish', there is always the risk that anglers may not always dispose of 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. Additionally, many Australian rivers are too turbid to be practical for bow-fishing.

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 alien fish species. Redfin perch are still highly sought after as recreational species in Australia, and roach and tench are still targeted by some coarse fishing enthusiasts in southern states. If recreational fishing for alien species is to be an activity supported by resource management agencies, then programs may need to be put in place to educate anglers about humane ways of capturing and disposing of 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.). Electrofishing from boats is generally constrained to waters less than 3 m deep and is a potentially useful method for reducing pest fish. Control of goldfish (*Carassius auratus*) is being undertaken in the Vasse River, Western Australia using a boat-mounted electric fishing machine (Morgan and Beatty 2006) and repeat use of this method can be successful in shallow water environments. Repeat electric fishing, using back-pack mounted machines, has also been used in small streams 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 where water depths exceed machine capabilities and where instream cover provides refugia from electrofishing.

Explosives were used by the New South Wales DPI in an attempt to eradicate a population of Jack Dempsy in a pool of a disused quarry in Angourie (Mick Holloway, NSW Fisheries, pers. comm.; ASFB 2006). They were also used in the Naroona Dam (Lake Navarino) in Western Australia to remove redfin perch (Molony et al. 2005). Although large numbers of perch were killed, they were not eliminated. Another application was in Bennet Brook in Western Australia to control a cichlid (*Geophagus braziliensis*) (pers. comm., D. Morgan). Explosives can be useful in small water-bodies where the 'effective' blast field can encompass the entire water mass. However, explosives have not proved effective in large, deep water bodies (Pullan 1982). This is because the 'effective' blast field is spatially limited and in large water bodies it may be impracticable to set enough charges to provide complete coverage. Even the extensive cover provided by the use of detonation cord and power gel explosives in the Angouri quarry may not have eliminated the Jack Dempsey cichlid, as this species has been found at the location subsequently (although this may have been due to a secondary introduction). Additionally, explosives are likely to be suitable for only very specific locations due to the resultant damage to aquatic habitats and other existing fauna.

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 (ABC 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; evaporation then resulted in desiccation and the removal of all fish.

Redfin perch were successfully removed from three reservoirs (the Pinwernying Dam, Phillips Creek Reservoir and Bottle Creek Reservoir) in Western Australia by draining them, and native fish have recolonised the Phillips Creek Reservoir (pers. comm. D. Morgan)

As noted above, 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.

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 alien pest fish species.

8.3 Chemical toxicants

Rotenone: The use of rotenone for the control of 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 1970s, and 1300 dams in Gippsland, Victoria in the early 1960s to control carp. Both programmes were considered successful, though carp were re-introduced to the Tasmanian dams some 20 years later and carp were recorded some 3 years later in the Yallourn storage dam in the La Trobe river system.

Rotenone was also applied unsuccessfully to ponds in Townsville to rid them of Mosambique tilapia (*Oreochromis mossambicus*) (Arthington et al. 1984), and to two ponds in residential properties in the Northern Territory to remove populations of gambusia (ASFB 2003a). In NSW, several rotenone treatments were needed to eradicate a population of one-spot livebearers from a series of ponds located on the

Long Reef Golf Course (Rayner and Creese 2006; ASFB 2008). In their review of rotenone use in Australia, Rayner and 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 a 2 ha pond near Ipswich in Queensland, redfin perch from Brushy Lagoon in Tasmania, and brown trout (*Salmo trutta*) 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 2003b). However, although rotenone was used to control gambusia in a farm dam in the Tamar River catchment of Tasmania in 1991, its population had re-established by 2001 and it had spread to two adjacent locations (Inland Fisheries Service 2007). In 2007, rotenone was used to eradicate perch from two dams in Tasmania. This treatment involved pumping and spraying to ensure complete coverage of all wetted areas as well as good mixing and was successful (ASFB 2007). Two dams in Tasmania were then treated with rotenone to eradicate gambusia using the same protocols as used for the successful perch removal and appear to have been successful (ASFB 2007).

Rotenone application is a highly effective method for the eradication of pest fish in enclosed systems, but local conditions have a large bearing on its success rate (Rayner and Creese 2006). In deciding whether to apply this chemical, the relevant agency needs to take into account the maximum depth of the water body, low water temperatures, flows, high turbidity, and exposure to sunlight.[10] Rotenoning is more viable in easily mixed,[11] shallow waterbodies 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. Another strategy is to apply rotenone when water levels are low, to minimize the spread of these chemicals or the need for neutralisation agents to be applied.

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[10] At water temperatures less than 12°C, rotenone use is less effective. Rotenone breaks down quickly under normal conditions, so its effects aren’t likely to be persistent. However, under low sunlight levels it can remain toxic for weeks (Sanger and Koehn 1997).

[11] 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 into water columns of shallow closed systems (see McDowall 2006).

The application of rotenone can result in collateral damage to native species unless salvage and resuscitation operations are carried out concurrently.[12] Fish resuscitation is possible by placing affected fish in clean water. The rotenone can also be neutralized by the addition of potassium permanganate to the water. However, 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.[13] At present, there have been reported links between the use of rotenone and the development of Parkinson's disease, and as such it does pose a human health risk (Betarbet et al. 2000; Giasson and Lee, 2000). However, the results of the study suggest that brain lesion symptoms in rats only occurred after prolonged exposure through blood infused directly with rotenone. It is strongly recommended that users follow the instructions set out on the label.

Rotenone was banned in Victoria (ASFB 2005). However, a permit has been procured for the use of rotenone in all states and territories, through the efforts of NSW DPI (Bob Creese, NSW DPI, pers. comm.) Legislation in New Zealand now prevents the use of the liquid form as it contains a synergist, the impacts of which are yet to be determined. The powder form (derris dust) is now used in New Zealand to avoid introducing chemical synergists into waterways.

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.[14] Rotenone has generally been applied over small areas, though there have been notable exceptions to this in other countries.[15]

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 2004). Sanger and Koehn (1997) have therefore

[12] This can be reduced, however, 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 and Koehn 1997).
[13] McDowall (2006) notes that many are similar to those used in household solvent products.
[14] 20 tonnes was used in a single reservoir in Utah (cited in McDowall 2006).
[15] A 400km stretch of river in Russia and a 700 km section of river in California were treated with rotenone (cited in McDowall 2006).

advocated that robust risk assessments and communication plans are prepared before rotenone is applied, with contingencies for emergency eradication situations. 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 as the patent holder may not approve this.

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 was registered as a piscicide in Australia as of 1996 (Sanger and Koehn 1997). The dose rates also require further clarification (ibid.). 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 rotenone (ibid.), 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 advantages over rotenone are 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 and 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 1960s. 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. 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 stormwater 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 two 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 utilized 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.

8.4 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, for example the use of Murray cod and shortfin eels to control common carp in the Glenelg River (ASFB 2004), and for the restoration of native piscivores to the upper reaches of rivers where aquarium fish now occur in degraded habitats (ASFB 2003b). But there are few instances where this has occurred. Australian bass were introduced to a waterway in New South Wales to control a wild population of Jack Dempsey. The costs involved in the sourcing of the introduced predator were not high, as the bass were being bred in the agency’s hatchery. Bass were also prevalent in the geographical location of the interaction (Mick Holloway, NSW Fisheries, pers. comm.) so escapees were not an issue.

To be effective, piscivores known to consume the target species, or at least to be capable of feeding on that species, need to be identified. In addition, the effectiveness of piscivorous fish will be governed by the degree to which the target pest fish species exhibits anti-predator behaviour,[16] how fast it can reproduce (i.e., how resilient its populations are likely to be to mortality through predation), the abundance of alternative prey species, and the prevalence of refugia for the prey species. Species of alien fish that exhibit anti-predator behaviour, 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. Gambusia are susceptible to predation by bass in water storages and perched coastal heathland lakes (A. Moore, pers. obs.). Density of gambusia in both cases substantially decreased after the introduction of bass and gambusia were no longer found in open water, being predominantly confined to macrophyte zones. Similar observations have been made with respect to the effect of trout and/or perch on gambusia (Rowe 2003; Molony et al. 2005). Although further research is needed to clarify the role of piscivores for the control of gambusia, there remains strong anecdotal evidence that it is likely to play an important role in suppressing gambusia populations. In an experimental evaluation of the response of gambusia to smallmouth bass (*Micropterus dolomieu*), Rehage et al. (2005a; 2005b) found that gambusia decreased its consumption rates and activity levels while increasing refuge use. This indicates that the presence of a fish predator can affect the behaviour and distribution of gambusia if not its abundance, and this too may modify its impact on native fish.

Choosing a predator species that is likely to be both effective for the purpose of its introduction and low risk in terms of potential ecological impacts, necessitates the need to use native fish rather than introduced species; though the current stocking program for salmonids in Australia may play an important role. Australia does not have many large, native, piscivorous predators (Koehn 2004), although it does have some species, currently being captively bred, that could be made readily available for control programs.

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 predator 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 alien fish species in closed systems. Whereas piscivore stocking may prove to be useful for gambusia control in closed environments, stringent risk

[16] There are several species that exhibit anti-predator behaviour including schooling, hiding and responding 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.

management plans, not unlike those put forward for rotenone use by Sanger and Koehn (1997), should be put in place whenever this is considered.

In North America, the red swamp crayfish (*Procambrus clarkii*) is a natural predator of gambusia, which appeared to show no anti-predator behaviour to it (Leite et al. 2005). Similarly, the yabbie (*Cherax destructor*) and marron (*Cherax cainii*) have been found to consume gambusia in the Hutt River in Australia (Beatty 2006). Large crayfish may therefore be considered as potential 'predators' of gambusia. The dynamics of crayfish-gambusia interactions clearly need to be explored further to see whether crayfish can be used to control gambusia populations and so reduce impacts.

Introduction of pathogens: The introduction of fish pathogens (e.g., parasites, bacteria, viruses) as a means of controlling or eradicating pest fish species is another method that is considered experimental rather than proven. Fish pathogens are usually specific to a family or even a genus of fish, so this technique can potentially be targeted at the pest species and not other fish.

In Australia, the introduced epizootic haematopoietic necrosis virus (EHNV) kills redfin perch (Langdon and Humphrey 1987) and has caused high mortality in some wild redfin perch populations. The introduction of the spring viraemia of carp virus (*Rhabdovirus carpio*) to Australia for carp control has been discussed since the 1970s (Crane and Eaton 1996), but this control method has not, to our knowledge, been implemented here due to concerns raised below. Carp herpes virus (CHV) is reported to kill four out of every five carp it affects in Europe and Asia (Pearson 2004) so whereas the spread of the virus is being actively prevented in the Northern Hemisphere, it may be a potential control agent in Australia where carp are a pest species. CSIRO are presently conducting trials to determine the effectiveness of this virus in eradicating carp in Australia and the potential for infection of native fauna.

Mass mortalities of gambusia (80% of the total population) occurred in an Indian lake in response to infection by an unidentified pathogen (Nagdali and Gupta 2002). Clearly, there are some pathogens that attack this species and which may not affect other species as much. However, research would be required to identify these and more importantly to rule out effects on other natural components of the fauna.

Fishes that live in harsh conditions and that are stressed are more likely to be susceptible to the impacts of pathogens. The effectiveness of pathogens will also be governed by environmental conditions (such as temperature) and might depend on the availability of intermediate hosts. Some viruses can be biochemically modified to be made more virulent, more or less host-specific, or to withstand a greater range of temperatures (Crane and Eaton 1996).

The effects of introduced pathogens on the host species are likely to decline as its populations become more resistant and/or resilient. Effectiveness will also depend on whether or not established alien fish populations are immunologically naïve to the pathogen in question. If they are, then introduced pathogens are likely to be much more effective. It may be difficult to assess whether or not this is the case for different wild populations in Australia before deciding whether this techniques is feasible. One of the main arguments against the potential effectiveness of this method will be that pathogens, even if they are initially effective, may become ineffective as the host population gradually acquires immunity to the pathogen.

At least 23 'natural' pathogens have been reported for gambusia in North America (Lloyd 1987, cited in Arthington and Lloyd 1989) and some of these could suppress gambusia populations in some places. As *Gambusia affinis* infected with the parasitic nematode worm *Eustrongyloides ignotus* were more susceptible to predation than non-infected fish (Coyner et al. 2001), parasite infestation could increase susceptibility to other control methods. EHN virus is known to significantly affect redfin populations (Langdon and Humphrey 1987). Key pathogens for roach, tench and gobies are not well studied in Australia, though these fish are likely to be susceptible to generalist aquatic pathogens.

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 risk 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 this would make sales of fish to other countries more 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 and this may prevent the introduction of pathogenic organisms to test their effectiveness as control agents.

[17] This would probably be the case if the spring viraemia of carp virus were introduced into Australia for carp control (Crane and Eaton, 1996).

8.5 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 alien fish species covered in this report.

To our knowledge, this method has not been applied yet in Australia, nor overseas, but is considered potentially viable because the populations of some freshwater fish that spawn in shallow waters on lake shores have declined following a reduction in water level (e.g., Gafny et al. 1992). Water level manipulation is currently being tested for carp control in shallow waters of the Barmah-Millewa forest (Gilligan 2005). This technique is also only likely to be viable where spawning habitats can either be altered or removed easily, or where it is practical to restrict the spawning migrations of established alien 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; this reinforces the need for more data on the habitat requirements of many of the established alien fish before this technique can be considered. Arthington et al. (1983), Webb (1994) and Kennard et al. (2005) all found that a number of alien fish species in northern Queensland were thriving in waters where degradation of the habitat had occurred through urban development. Development in this context included removal of riparian trees, increased siltation of substrates, and increased nutrient inputs, all of which served to expose streams to increased macrophyte growth and stagnation, which disadvantaged native fish but assisted the survival of alien fish. As a consequence, Webb (1994) 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 aquarium species (c.f., Arthington et al. 1990). Bunn & Arthington (2002) noted that gambusia were reduced by floods and Pritchard et al. (2004) have advocated flood 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

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[18] Several species of established alien fish do have certain requirements for spawning, including the need for fish passage during migration, and specific substrates (e.g., *Tilapia mariae*). Populations of these species may be able to be controlled to a degree using this control method.

disconnected, isolated pools favouring gambusia and increased their exposure to native piscivores. Gambusia do not cope well with increased water velocities and floods have greatly reduced their populations in some Australian rivers (Meffe 1984; Pusey et al. 1989; Chapman & Warburton 2006). This indicates that the maintenance of flood flows, especially in summer, is important for keeping gambusia in check. Increasing water velocities in stream channels will also be a useful tool to reduce their habitat. 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.

Lake (1967a) noted that the survival of redfin perch eggs was greatly reduced if water temperatures rose too quickly because this resulted in early hatching at a premature size and state. He also noted that egg deposition was strongly influenced by substrate. In ponds covered with a silty substrate and lacking either submerged macrophytes or woody debris the eggs were deposited on shallow grassy margins. This specificity of spawning substrate and the dependence of egg survival on low water temperatures may offer scope for habitat modification to reduce perch reproduction in some ponds and lakes.

8.6 Immuno-contraceptive control and genetic techniques

As with biological controls, these methods are also considered to be experimental rather than proven techniques. While both techniques have the potential to reduce populations of pest fish species through a reduction in their reproductive output, reductions in fertility can sometimes be compensated for by greater survivorship of juveniles through lower levels of intra-specific competition. Thus, a high level of fertility reduction over time may be required before any major effects on abundance are realised (Hinds and Pech 1996).

Chemicals with estrogenic activity have been found to shorten the gonopodium of male gambusia (e.g., Batty and Lim 1999; Game et al. 2006). This phenomena occurs in rivers downstream of treated sewage discharges from major urban centres and although synthetic estrogens used for human contraception are involved, a number of other chemical agents (e.g., alkylphenols, and chlorinated hydrocarbons such as endosulphan, dieldrin, PCB) are also likely to be responsible (Batty and Lim 1999). Although gambusia is studied as a test organism to detect the presence of such chemical pollutants, there has been little work done to determine the consequences for its reproduction and hence for population control (Game et al. 2006). Gambusia may be more susceptible to such effects than other fish species. If so, it is one aspect of immuno-contraceptive control that needs to be examined in more depth to see whether there is scope for gambusia control.

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 are 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 focused on the development of a 'daughterless carp' gene in Australia (Ron Thresher, CSIRO, pers. comm.). However, attempts to introduce such a gene into *Gambusia* to demonstrate the viability of the method were not successful, so its application to all some alien fish may be problematic. 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 and Koehn (1997), should be put in place whenever this method is considered.

8.7 Alternative species

Gambusia was imported to many countries around the world for the control of mosquito larvae and has few other uses, apart from this. The provision of alternative methods of mosquito larvae control in Australia can be expected to greatly reduce its illegal spread. In this respect a number of studies have now evaluated the effectiveness of a range of native fish species for mosquito larvae control (Lloyd 1986; 1989; Morton et al.1988; Willems et al. 2005; Hurst et al. 2006). A number of native fish species were found to be more effective at preying on mosquito larvae and these are now routinely recommended by state agencies for mosquito control in waterbodies such as ponds and small lakes (e.g., Queensland DPI 2006).

8.8 Summary of control and eradication options:

There are a wide range of potential options for the control and/or eradication of established alien fish species, 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 waterbodies 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 alien 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 in place. There also need to be stringent risk assessments and communication plans developed for many of these control and eradication techniques. We note that this is something that has been considered as part of the Operational Strategy for Control of Exotic Fishes in Queensland (MacKenzie 2003).

The effectiveness of control and eradication programs can only be quantified if rigorous monitoring programs are put in place that will allow before and after treatment densities of the target species to be determined, and/or a reduction in impacts to be measured. This will require the use of pilot studies to determine the adequate number of samples required to detect a change between treatments and controls. In association with these programmes, it may be desirable to monitor changes in both populations of the targeted species and those of certain native fish species. This is for the reason 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

9. Policy and regulatory framework

9.1 Introduction

Alien fish have been introduced into inland Australian waters since the mid 19th century for various reasons: as ornamental species, for recreational and commercial fishing, and to control various pests. According to McKay (1984), the introduction of alien fish into Australia can be broadly categorised into three periods. The first period, between 1862 and 1896, involved the introduction of at least 10 species to create a familiar environment for the European settlers. The second period, initially during the 1920s and again during the Second World War, involved the release of gambusia to control mosquito populations. Finally, the more recent introductions during the 20th century have generally been for aquaculture and aquarium purposes. With the exception of the yellowfin and streaked gobies, all of the alien fish within Australia's inland waters have been deliberately introduced. There are currently forty two alien freshwater fish species established in freshwater environments in Australia.[19] Many of these species continue to flourish, and while some research has been undertaken, and a number of eradication programs established (especially targeting carp), there is limited information on the prevalence and impact of many of them.

The control of alien fish is governed by commonwealth as well as state and territory governments and the associated federal, state and local policies. There are numerous pieces of legislation and various policies relating to the control of alien fish. The legislation ranges from fairly broad commonwealth acts and regulations to more specific state acts and regulations. Historically, control over alien fish was based primarily on legislation covering fisheries. Legislation for maintaining biosecurity added another layer to this and, more recently, concerns over the decline of native species has resulted in a third layer of threatened species legislation.

The fisheries legislation included provisions to list alien fish as noxious or prohibited species and made it illegal for anglers to deal with these fish in any way not allowed by the legislation (i.e., if caught by anglers, such fish had to be killed and could not be returned to the water alive). This legislation was designed to stop the spread of such species by anglers and jurisdictions using best practice (e.g., Northern Territory) developed list for both permitted and prohibited fish species).

The initial legislation covering biosecurity was designed primarily to stop the spread of diseases and parasites that could affect primary industries or human health. Today, a more comprehensive and integrated approach to biosecurity is required that encompasses invasive species and their ecological impacts as well as animal health threats. Such an approach has been adopted by Western Australia's Biosecurity and Agriculture Management Act, 2007. Other states such as Queensland and Victoria are

[19]Lintermans (2004) listed 34 species, excluding 8 additional species listed in Corfield et al. (2008). These reports collectively identify 42 alien species that have established in the wild.

now consolidating their administration of biosecurity through the formation of new state policy and agencies (e.g., Biosecurity Queensland and Victoria, respectively).

Threatened species legislation is designed to halt the decline of Australia's endemic flora and fauna, especially rare species that are under threat because of changes caused by human activity. The Environment Protection and Biodiversity Conservation Amendment (Wildlife Protection) Act 2001 repealed the Wildlife Protection (Regulation of Exports and Imports) Act 1982 and incorporated an enhanced wildlife protection regime into the Environment Protection and Biodiversity Conservation Act 1999. The Commonwealth Endangered Species Protection Act 1992 was repealed and the prohibited fish list in Schedule 6 of the Wildlife Protection Act 1982 was incorporated into the Environment Protection and Biodiversity Conservation Act. The Department of Environment and Water established a list of organisms suitable for live import into Australia and the Australian Government quarantine legislation is applied separately to that of the Department.

Threatened species legislation in the states and territories allows the listing of 'key threatening processes (KTP)' for rare and endangered endemic species. Such processes may be specific (i.e., gambusia is a KTP in New South Wales), or general (i.e., the deliberate or accidental introduction of live fish outside their natural distribution is a KTP in Victoria). KTP's may also be listed by the commonwealth under the Environment Protection and Biodiversity Conservation Act 1999.

Alongside each piece of legislation, there are numerous state policies. At a local level some specific regions have their own management plans such as the Western Lakes - Fishery Management Plan 2002 (TAS). To date, Queensland is the only state to have developed a multi-species pest management strategy for alien freshwater fish species.

The legislation dealing with alien freshwater fish species was first reviewed by Wager and Jackson (1993) and Wager (1994), and later updated by Arthington and McKenzie (1997). A review of the specific implications of this legislation as it relates to alien fish in Queensland was later provided by the Queensland Department of Primary Industry (QDPI 2001).

This chapter provides a broad summary of the legislation and policies which seek to guide the control of pest fish species, including those of interest to this study (gambusia, redfin perch, roach, tench, yellowfin goby and streaked goby). It focuses on the main commonwealth and state acts and policies but is by no means an exhaustive list of all the Acts and policies which could apply to the control of pest species. Rather, it aims to provide an overview of the existing legislation and policy in order to identify the main provisions and any major gaps that may need to be addressed.

9.2 Analysis of Commonwealth and State legislation

The main federal policies and acts and the various state acts and regulations influencing the management of alien fish species in Australia are shown in Figure 9.1.

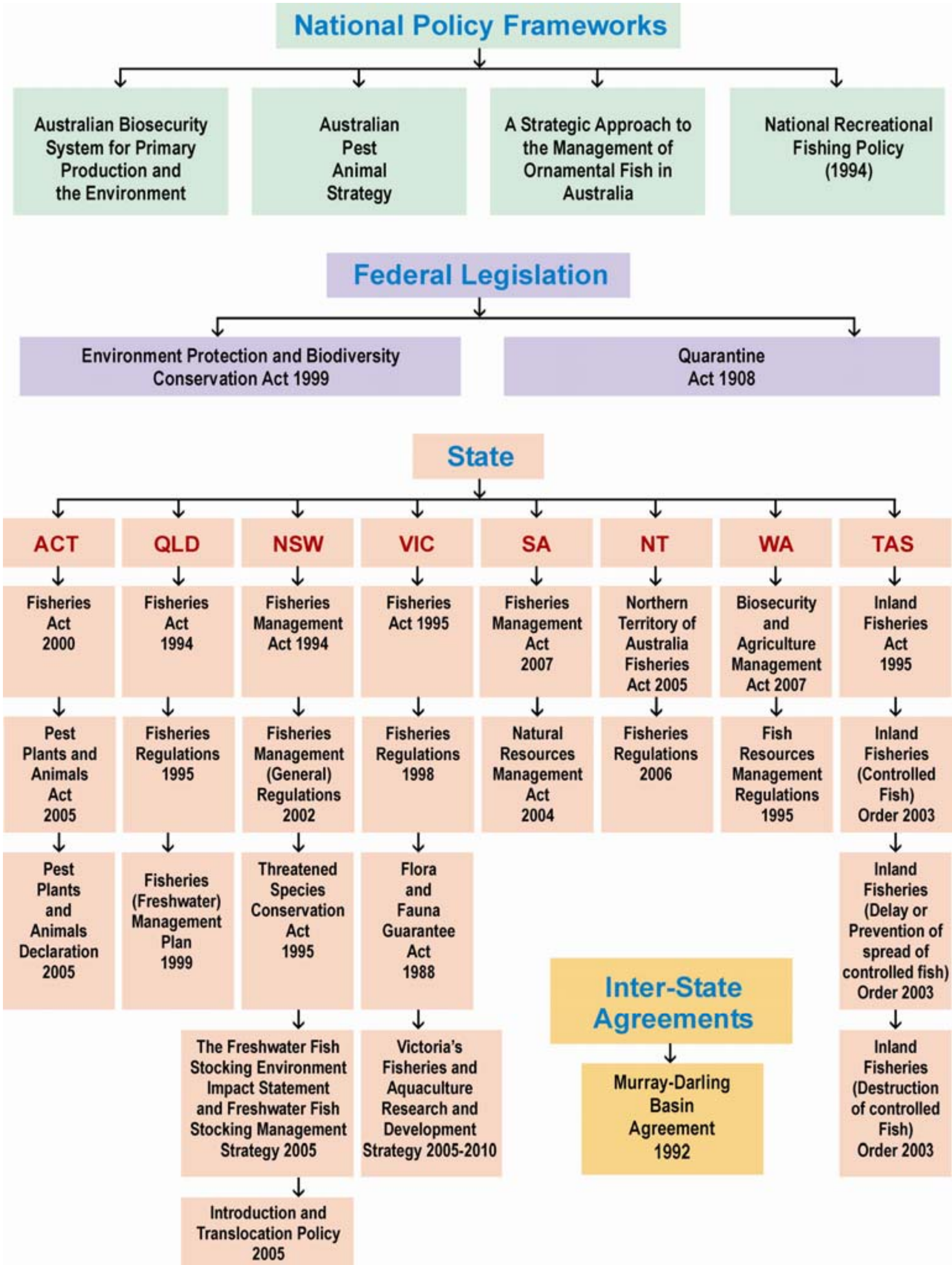

Figure 9.1: Overview of the federal and state legislation influencing the management of alien freshwater pest fish species in Australia.

The federal acts deal with the importation of commodities to minimise the risk of exotic pests and diseases entering the country (Quarantine Act 1908) and with the protection of the environment and its fauna (Environment Protection and Biodiversity Conservation Act 1999). State legislation is focused more on the management of commercial and recreational fish and fisheries (Fisheries Acts), with legislation also present in some states for environmental protection (including threatened species) and/or the management of pest species. Other legislation is designed to manage aquaculture and disease spread via stocking. A description of the objectives and main clauses relating to alien fish management for each of the main pieces of legislation is provided in Table 9.1.

Table 9.1: Analysis of federal and state policy and legislation influencing the management of the species reviewed in this report.

Name of Policy/Regulation	Key Objectives and Relevant Clauses	Analysis
(A) National Policies		
AusBIOSEC (Australian Biosecurity System for Primary Production and the Environment)	AusBIOSEC (Australian Biosecurity System for Primary Production and the Environment) is a framework of common principles and guidelines to enable biosecurity arrangements to be applied consistently across Australia. The aim is to bring together all biosecurity activities being undertaken by the Australian Government, state and territory governments, industry, landholders and other key stakeholders in primary production and the environment. AusBIOSEC covers all invasive plants, animals and diseases, of the terrestrial and aquatic environment that could be harmful to primary industries, the natural and built environments, and public health. The joint steering group for AusBIOSEC includes representation from the Australian Government; State and territory primary industries, environment and natural resource management agencies; Local Government, and the CSIRO. It reports to the Natural Resource Management Ministerial Council and the Primary Industries Ministerial Council through their respective Standing Committees. A joint DAFF/DEW Secretariat (located in DAFF) has been established to support the Joint Steering Group and its working groups.	AusBIOSEC is the emerging nationally coordinated biosecurity regime. The current focus by the Natural Resources Management and Primary Industries Ministerial Councils includes the endorsement, in-principle to an Inter-Governmental Agreement (IGA) for AusBIOSEC. The IGA will define clearly the responsibilities of the Australian and state and territory governments in improving Australia's biosecurity system. National cost sharing arrangements for the public good component of a national emergency response to an incursion is an issue also under consideration. A national response would first require agreement that the incursion is of a nationally significant invasive species; the response would be feasible and cost-effective; and the incursion is not eligible for funding under other emergency response arrangements which involve industry cost-sharing. Council gave in-principle agreement to a cost sharing formula which would be based on the potential of an invasive species to affect a jurisdiction and the principle that beneficiaries contribute to a response. [20]

[20] See AusBIOSEC overview at: http://news.envirocentre.com.au/lawn/newsletterfull_.php?issue=2006-12-06&key=5#2225

Australian Pest Animal Strategy	The Australian Pest Animal Strategy is a vital part of Australia's integrated approach to national biosecurity under the Australian Biosecurity System for Primary Production and the Environment (AusBIOSEC). It complements existing and new strategies, covering weeds, marine pests and animal welfare.[21] The Australian Pest Animal Strategy identifies the following goals and objectives to realise its vision: Goal 1: Provide leadership and coordination for the management of pest animals Objective 1.1 To develop the capacity and processes for effective delivery of pest animal management. Objective 1.2 To ensure nationally consistent pest animal management approaches are in place at all scales of management. Objective 1.3 To improve public awareness of pest animals, research coordination and its support for pest management at the national level, and adoption of best practice management methods. Goal 2 Prevent establishment of new pest animals Objective 2.1 To prevent the introduction of new animals with pest potential. Objective 2.2 To ensure early detection of, and rapid response to, new incursions of exotic animals. Objective 2.3 To reduce the spread of pest animals to new areas within Australia. Goal 3 Manage the impacts of established pest animals Objective 3.1 To identify established pest animals of national significance. Objective 3.2 To identify and manage the impacts of pest animals on key assets. Objective 3.3 To coordinate the management of established pest animals across Australia. These objectives will be implemented through a set of strategic actions, with progress to be measured against specific outcomes.	The APAS is the first national pest animal policy framework developed by Commonwealth, State and Territory governments. It was prepared by the Vertebrate Pests Committee and adopted by the Natural Resources Management Ministerial Council in April 2007. It mirrors the Australian Weeds Strategy, and similarly nests under the emerging AusBIOSEC. Its features include a focus on pest spread pathways, protection of natural assets of national importance, and the identification of a robust list of pest animals of national importance.

[21] http://www.environment.gov.au/biodiversity/invasive/publications/pubs/pest-animal-strategy.pdf

A Strategic Approach to the Management of Ornamental Fish in Australia (NRMMC 2006)	The key objectives for the development of a nationally agreed approach on aquarium fish addressed in this plan are: • The development of a strategic plan for management for ornamental fish • The development of a national list of high-risk noxious species • The development of a national exempt (low risk) list of species permitted in the ornamental fish trade • A process for assessing the risks associated with any species currently in Australia that may not have previously been assessed • A process for dealing with undesirable species already in the country • Consultation with stakeholder groups on the implementation of changes. The National Strategy proposes a number of national policy and legislative innovations, including development and implementation of uniform national invasive species control classes, and a national noxious fish list. Three national control classes are set out in the Federal Senate Inquiry on Invasive Species. • National Quarantine List: Invasive species of national importance that are a high invasion risk for Australia and not known to be present in Australia, and whose early detection will enable cost effective eradication • National Alert List: Invasive Species of national importance that are naturalised, have a restricted range, are predicted to have a major impact on the environment, human welfare or industries, and which may be, is currently, or was, subject to a State or national eradication effort • National Control List: Invasive species of national importance that are naturalised and generally widespread, are having a major impact on the environment or industry, and whose containment or control will assist protect the values of areas of national environmental significance. Also see Australian Biosecurity Group (2005)[22] The Strategic Plan implementation is focussed on the: • development and adoption of a national noxious species list across all jurisdictions, noting links to existing lists and lists under development. • Review of the status of fish on the 'grey list' as a national priority • Establishment of a scientific/technical working group to conduct assessments of fish on the grey list over the next 2-3 years • Adoption of a regulatory framework and licensing to manage large fish-breeders and ornamental fish importers in each state and territory • Development of control mechanisms for the regulation and management of noxious fish and rare fish (e.g., CITES listed) already in circulation in Australia, again noting links to control plans for marine pests of concern. • Initiation of a review of aquatic plants used in the ornamental fish trade, in order to control and regulate the spread of recognised pest species • Implementation of a national communication strategy to raise awareness about the management, control and regulation of ornamental fish.	*A Strategic Approach to the Management of Ornamental Fish in Australia* is a national policy adopted by the Natural Resource Management Ministerial Council in November 2006. It nests under the Australian Pest Animal Strategy and the Australian Weeds Strategy. Key issues addressed by the Strategy include: • The large number of ornamental species in the country that are not on the live import list under the EPBC Act • The disease and pest status of animals that may have entered the country illegally • Inconsistencies between jurisdictions in legislation and policy relating to permitted/noxious species and effective controls • The effectiveness of current border controls to prevent illegal imports of species and consequent potential animal health risks The element of the Strategic Plan of most relevance to the scope of this study is the proposed national noxious fish list. Two of the six pest fish species that are the focus of this study are included in the pest fish species agreed by all jurisdictions for inclusion on the national noxious fish list: These are yellow fin goby (*Acanthogobius flavimanus*) and gambusia (*Gambusia holbrooki*). A further three pest fish species have been agreed by all jurisdictions for inclusion on the national grey fish list, which requires further scientific and technical consideration and risk assessment: tench (*Tinca tinca*), roach (*Rutilus rutilus*), and redfin perch (*Perca fluviatilis*). The only pest fish reviewed in this study that is not on the national noxious nor grey lists is the streaked goby (*Acentrogobius pflaumi*).

[22] http://www.wwf.org.au/publications/ABGInvasiveSolutions/

National Recreational Fishing Policy (1994)	As part of its 2007 election *Plan for Sustainable Fisheries*, the Australian Labour Government has committed to review the 1994 National Recreational Fishing Policy, and prepare a new Recreational Fishing Industry Development Strategy to encourage sustainable fishing. The Recreational Fishing Industry Development Strategy will provide opportunities for the recreational fishing industry to: • Provide effective national representation of the recreational fishing industry including the state and territory peak bodies and the fishing tackle sector. • Advise the Minister on the review of the 1994 National Recreational Fishing Policy and development of the Recreational Fishing Industry Development Policy. • Advise the Minister on specific initiatives to encourage the promotion of sustainable fishing, education and awareness for children and angler communities, and best practice environmental standards for recreational fishing. The Australian Government will work closely with Fisheries Research and Development Corporation to invest in leadership training and education and awareness campaigns for the recreational fishing industry (ALP 2007, pg.9)[23] Five primary goals and sixteen guiding principles form the basis of The National Recreational Fishing Policy. Those relevant to this report are: • Recreational fishing should be managed as part of the total fisheries resource to ensure quality fishing, and to maintain fish stocks and their habitats, for present and future generations of Australians. • Our aquatic habitats and ecosystems are part of the environmental endowment of all Australians, and are the key to a healthy fisheries resource which requires protection, restoration and enhancement. • Government, in its stewardship role, must encourage and assist the community to be involved in all aspects of fisheries management. • Recreational fishers and the recreational fishing industry should participate in the protection and management of their fishing heritage to ensure that it is available for future generations. • Community consultation at Federal, State/Territory and local levels should be a key component of recreational fisheries management programs. • Community awareness, education and enforcement programs should focus on encouraging positive changes in community attitudes to develop a stronger conservation ethic.	In the lead up to the November 2007, Federal election, the ALP committed to invest $2M to develop a new Recreational Fishing Industry Development Strategy under it's *Plan for Sustainable Fisheries*.[24] The policy does not distinguish between native and non-indigenous species of fish and does not specifically refer to any of the fish species of interest in this study. However it does refer to the need to maintain or enhance fish stocks, which process is regulated by the Fisheries Act for each State. In addition, it encourages community awareness through positive changes in attitude which may reduce the likelihood of people desiring pest fish species for the purpose of recreational fishing.

[23] http://www.alp.org.au/download/now/071120_fisheries_policy.pdf
[24] http://www.alp.org.au/download/now/071120_fisheries_policy.pdf

	• The economic, educational, health and other social benefits of recreational fishing should be widely recognised and actively promoted. • Fisheries management decisions should be based on sound information including fish biology, fishing activity, catches, and the economic and social values of recreational fishing. Recreational fishers should continue to contribute to the cost of managing and developing recreational fishing. • • Ensure quality fishing, and maintain or enhance fish stocks and their habitats, for present and future generations as part of the environmental endowment of all Australians. • Develop partnerships between governments, the recreational fishing community, and associated industries to conserve, restore and enhance the values of recreational fisheries throughout Australia • Establish an information base at national and regional levels to meet the needs of recreational fisheries management. • Establish a funding base to effectively manage the nation's recreational fisheries. community, and associated industries to conserve, restore and enhance the values of recreational fisheries throughout Australia • Allocate a fair and reasonable share of Australian fish resources to recreational fishers, taking into account the needs of other user groups. • Recreational fishing should be managed as part of the total fisheries resource to ensure quality fishing, and to maintain fish stocks and their habitats, for present and future generations of Australians. • Our aquatic habitats and ecosystems are part of the environmental endowment of all Australians, and are the key to a healthy fisheries resource which requires protection, restoration and enhancement. • Government, in its stewardship role, must encourage and assist the community to be involved in all aspects of fisheries management. • Recreational fishers and the recreational fishing industry should participate in the protection and management of their fishing heritage to ensure that it is available for future generations. • Community consultation at Federal, State/Territory and local levels should be a key component of recreational fisheries management programs.	

(B) Federal Legislation	Key objectives	Analysis
Environment Protection and Biodiversity Conservation Act 1999	The *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act) protects the environment, particularly matters of National Environmental Significance. It streamlines national environmental assessment and approvals process, protects Australian biodiversity and integrates management of important natural and cultural places. The EPBC Act came into force on 16 July 2000.[25] The relevant objects of this Act are: (a) to provide for the protection of the environment, especially those aspects of the environment that are matters of national environmental significance; (b) to promote ecologically sustainable development through the conservation and ecologically sustainable use of natural resources; (c) to promote the conservation of biodiversity; (d) to promote a co-operative approach to the protection and management of the environment involving governments, the community, land holders and indigenous peoples; (e) to assist in the co-operative implementation of Australia's international environmental responsibilities; (f) to recognise the role of indigenous people in the conservation and ecologically sustainable use of Australia's biodiversity; and (g) to promote the use of indigenous people's knowledge of biodiversity with the involvement of, and in co-operation with, the owners of the knowledge. In order to achieve its objects, the Act: (a) recognises an appropriate role for the Commonwealth in relation to the environment by focussing Commonwealth involvement on matters of national environmental significance and on Commonwealth actions and Commonwealth areas; (b) strengthens intergovernmental co-operation, and minimises duplication, through bilateral agreements; (c) provides for the intergovernmental accreditation of environmental assessment and approval processes; (d) adopts an efficient and timely Commonwealth environmental assessment and approval process that will ensure activities that are likely to have significant impacts on the environment are properly assessed; (e) enhances Australia's capacity to ensure the conservation of its biodiversity	The Act streamlines national environmental assessment and approval processes, protects Australian biodiversity, and integrates management of important natural and cultural places. It focuses on the integration of various stakeholders and, in particular, recognises the important role indigenous people play in the conservation of Australia's biodiversity. The Act attempts to consolidate various state legislation on environmental protection and biodiversity, to provide a unified approach to conservation. The Act does not specifically address pest species of fish other than through the live import list; however the broader objectives of the Act clearly discourage the spread of exotic species, with a focus on protecting and enhancing native species and their environments. It therefore provides a sound platform for managing Australia's natural resources and for protection from potentially invasive pests. It also emphasises the role of the community and indigenous communities in the management of natural resources. International import and export of live fish species: The current *List of Specimens taken to be Suitable for Live Import* is an artefact of Australia's previously less regulated quarantine arrangements, and includes several significant weaknesses or complicating factors including:

[25] Verbatim from: http://www.environment.gov.au/epbc/about/index.html
[26] The list is at: http://www.environment.gov.au/biodiversity/trade-use/lists/import/pubs/live-import-list.pdf
[27] Taken verbatim from: http://www.environment.gov.au/biodiversity/trade-use/lists/import/index.html
[28] http://scaleplus.law.gov.au/html/pasteact/3/3295/0/PA004970.htm

	by including provisions to: (i) protect native species (and in particular prevent the extinction, and promote the recovery, of threatened species) and ensure the conservation of migratory species; (iii) protect ecosystems by means that include the establishment and management of reserves, the recognition and protection of ecological communities and the promotion of off-reserve conservation measures; (iv) identify processes that threaten all levels of biodiversity and implement plans to address these processes; and (f) includes provision to enhance the protection, conservation and presentation of world heritage properties and the conservation and wise use of Ramsar wetlands of international importance; (g) includes provisions to identify places for inclusion in the National Heritage List and Commonwealth Heritage List and to enhance the protection, conservation and presentation of those places; and (h) promotes a partnership approach to environmental protection and biodiversity conservation through: (i) bilateral agreements with States and Territories; and (ii) conservation agreements with land holders; and (iii) recognising and promoting indigenous people's role in, and knowledge of, the conservation and ecologically sustainable use of biodiversity; and (iv) the involvement of the community in management planning. International import and export of live fish species: International movement of wildlife and wildlife products is regulated under Part 13A of the *Environment Protection and Biodiversity Conservation Act 1999* for all wildlife. The import of live plants and animals into Australia is specifically regulated under section 303(EB) of the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act). All species permitted for import into Australia are included on the list of specimens suitable for live import (the live import list).[26] Species not identified on this list cannot be legally imported into Australia.	• The list comprises a combination of species and genera. The genera enable the potential import of new invasive fish species with no risk assessment. A similar genera loophole for plants (the Schedule 5 permitted seeds list) under the *Quarantine Proclamation 1998*, was rectified by the Australian Government in Dec. 2006. • The list is not reflective of the non-native fish species already in Australia, though this more restrictive list is a strength of the new stronger biosecurity measures. For example, a national review of exotic ornamental fish in Australia and their legal status that showed that there were 1100 exotic ornamental fish but only 481 species or genera listed under Part 13A of the EPBC Act (McNee 2002).[29] Invasive species regulations: As of March 2008, the Commonwealth has not promulgated invasive species regulations under section 301A of the EPBC Act. They were, however, considered through a Federal Inquiry by the Senate Environment, Communications, Information Technology and the Arts Committee. Its findings were published in December 2004 as Turning the Tide – the invasive species challenge: Report on the regulation, control and management of invasive species and the Environment Protection and Biodiversity Conservation Amendment (Invasive Species) Bill 2002). The Committee recommended that the Australian Government promulgate regulations under

29 McNee, A. 2002. A national approach to the management of exotic fish species in the aquarium trade: An inventory of exotic freshwater species. Report for Fisheries Resources Research Fund. Bureau of Rural Sciences, Canberra.

30 http://www.aph.gov.au/Senate/Committee/ecita_ctte/invasive_species/report/index.htm

31 http://www.environment.gov.au/cgi-bin/sprat/public/publicgetkeythreats.pl

32 http://www.environment.gov.au/biodiversity/threatened/publications/pubs/priority-assessment-list.pdf

	There are two parts to the live import list. Part 1 contains species that can be brought into Australia without a permit from the Department of the Environment, Water, Heritage and the Arts (the Department). Part 2 contains species that require a permit from the Department before being permitted for import into Australia. Conditions may be attached to the permit.[27] The inclusion of a species on the live import list allows that species to be imported as either a whole organism or as reproductive material. Regulations for invasive species: Section 301A of the EPBC Act makes provisions for regulations to control non-native species. The regulations may: (a) provide for the establishment and maintenance of a list of species, other than native species, whose members: (i) do or may threaten biodiversity in the Australian jurisdiction; or (ii)would be likely to threaten biodiversity in the Australian jurisdiction if they were brought into the Australian jurisdiction; and (b) regulate or prohibit the bringing into the Australian jurisdiction of members of a species included in the list mentioned in paragraph (a); and (c) regulate or prohibit trade in members of a species included in the list mentioned in paragraph (a): (i) between Australia and another country; or (ii) between 2 States; or (iii) between 2 Territories; or (iv) between a State and a Territory; or (v) by a constitutional corporation; and (d) regulate and prohibit actions: (i) involving or affecting members of a species included in the list mentioned in paragraph (a); and (ii) whose regulation or prohibition is appropriate and adapted to give effect to Australia's obligations under an agreement with one or more other countries; and (e) provide for the making and implementation of plans to reduce, eliminate or prevent the impacts of members of species included in the list mentioned in paragraph (a) on biodiversity in the Australian jurisdiction. [28]	section 301A of the EPBC to prohibit the trade in invasive plant species of national importance, combined with State and Territory commitment to prohibit these same species under their respective laws.[30] In its response to the Committee's recommendations, the Australian Government disagreed with this particular recommendation (Recommendation 6), and noted that in the first instance, the States and Territories should strengthen the control and management of invasive species (specifically weeds) within their respective jurisdictions. Listing pest fish as Key Threatening Processes: The Act sets out a list of key threatening processes in Chapter 5, Division 1, Section 183. As of March 2008 no pest fish are listed as Key Threatening Processes under section 183 of the EPBC Act.[31] However, in 2007, the Minister accepted a *Finalised Priority Assessment List for the Assessment Period* commencing 1 October 2007, which requires the Threatened Species Scientific Committee to assess the public nomination of the introduction of live native or non-native fish into Australian watercourses that are outside their natural geographic distribution by 30 September 2009.[32] The Federal Threatened Species Scientific Committee is currently preparing a 'Finalised Priority Assessment List for the period commencing 1 October 2007' and is considering the inclusion of "the introduction of live native or non-native fish into Australian watercourses that are outside their natural geographic range" .
Quarantine Act 1908	Part V of the Act relates to the quarantine of animals and plants. It specifically states that: 1.The Governor-General may make regulations, not inconsistent with this Act, prescribing all matters which by this Act are required or permitted to be prescribed or which are necessary or convenient to be prescribed for carrying out or giving effect to this Act, and, without limiting the generality of the	This Act therefore controls the animals entering Australia, and applies specific controls on ballast waters of vessels. This Act aims to reduce the introduction of pest animals through careful examination and quarantine measures.

	foregoing, may make regulations concerning the following matters: [relevant clauses are listed in the order they appear in the Act] (f) for prescribing the precautions to be taken to prevent the ingress to or egress from a vessel of rats, mice, mosquitoes or other vermin or species or kinds of animals or insects liable to convey disease or pests; (g) for prescribing the measures to be taken by the masters or owners of vessels to destroy rats, mice, mosquitoes or other vermin or species or kinds of animals or insects liable to convey disease or pests, which may exist on the vessels; (h) for prescribing and for establishing and maintaining on vessels or within any quarantine area of conditions unfavourable to, and to the migration of, rats, mice, mosquitoes or other vermin or species or kinds of animals or insects liable to convey disease or pests, for fixing the time limit for the completion of any work necessary for the purpose of establishing such conditions, and for empowering the Minister, in case of default by the owner or master, to carry out any such work at the expense of the owner or master; (i) for prescribing the precautions to be taken by masters of vessels in respect of their vessels and the crews, passengers and cargoes of their vessels: to prevent the introduction into Australia, the Cocos Islands or Christmas Island, or the establishment or spread, of quarantinable diseases or quarantinable pests (k) for regulating the discharge or removal from a vessel of any thing, for example, ballast water, refuse, and equipment or things used for purposes associated with the transportation of animals, plants or other goods; (q) for prescribing methods of controlling the storage, use, movement and disposal in Australia, the Cocos Islands or Christmas Island of goods that: have been imported into Australia, the Cocos Islands or Christmas Island, being goods the importation of which without the approval of the Minister or a Director of Quarantine is prohibited by a proclamation under section 13 or the importation of which without a permit granted pursuant to a proclamation made in accordance with subsection 13(2A) is prohibited; or are disease agents or pests produced in Australia from goods of the kind referred to in subparagraph (i).	Although it doesn't specifically mention any particular species of animal or plant, it remains general to all introduced animals and plants. The term 'animal' includes dead animals and any part of an animal, including pest fish. Biosecurity Australia's Import Risk Analysis (IRA) determines pest and disease policies in relation to commodity imports as outlined in www.daff.gov.au/ba/ira

(C) State Acts and Regulations		Objectives	Analysis
ACT	Fisheries Act 2000	Section 14 Power to declare a fish species noxious Section 67 Power to seize and destroy noxious fish Sections 74-78 Prohibits possession, trafficking, importing and exporting of noxious fish without a permit, license or written approval from the Conservator . Prohibits the release of live fish into public waters without a conservators written approval	Provides powers to control the holding and movement of species declared to be noxious, and allows possession by written permit. Also prohibits release of live fish into public (but not private) waters without written permission.
	Pest Plants and Animals Act 2005	Section 16 Power to declare a species a pest Section 18 Requirement for owner/occupiers to provide notification of the presence of a pest Section 22 Prohibition on possession Section 23 Permits to supply pests Section 25 Power to manage pest species	Provides powers to identify pest species, to control their impact via permits on possession or supply, and allows for the Commissioner for the Environment to order management of the pest by the occupier of the place where it occurs.
	Pest Plants and Animals (Pest Animals) Declaration 2005 (1)	Lists pest species and identifies which are notifiable and prohibited.	Yellowfin goby is listed as a notifiable pest Gambusia is listed as a pest but is not notifiable or prohibited

QLD	Fisheries Management Act 1994	(1) The main purposes of this Act is to provide for the use, conservation and enhancement of the community's fisheries resources and fish habitats in a way that seeks to: (a) apply and balance the principles of ecologically sustainable development; and (b) promote ecologically sustainable development. (2) In balancing the principles, each principle is to be given relative emphasis appropriate in the circumstance. (3) Ecologically sustainable development means, development: (a) carried out in a way that maintains biodiversity and the ecological processes on which fisheries resources depend; and (b) that maintains and improves the total quality of present and future life. Section 89 of the Fisheries Act 1994 states: A person must not unlawfully – (a) bring noxious fisheries resources, or cause noxious fisheries resources to be brought into Queensland; or (b) possess, rear, sell or buy noxious fisheries resources; or (c) release noxious fisheries resources, or cause noxious fisheries resources to be places or released, into Queensland waters. Section 92 of the Act states that a person who unlawfully takes or possesses noxious or non-indigenous fisheries resources must kill it immediately It also defines *noxious fisheries resources* as fisheries resources which are prescribed under a regulation or management plan to be noxious fisheries resources.	This Act focuses on the appropriate management of fish and promoting the conservation/enhancement of fish stocks in a sustainable manner. It also specifically mentions that noxious fish species should not be brought into the State's waters. Of the species of interest to this study, the only fish species identified as noxious is gambusia.
	Fisheries Regulations 1995 (revised in 2008)	Schedule 5A lists the common names of eighteen noxious fish in Queensland and includes gambusia (mosquito fish). It does not list any of the other species of interest to this study. Section 10 of the Fisheries Regulation 1995 states the references used for noxious and exotic fish species. Under Section 35 of the Regulation, General Fisheries Permits are issued for stocking Queensland waters. Section 81 of the Fisheries Regulation 1995 states: A person may do the following things involving fisheries resources only if the person holds an authority for the purpose: (a) bring them or cause them to be brought into Queensland; (b) possess, rear, sell or buy them; (c) release them, or cause them to be released into Queensland waters.	The Fisheries Regulation is the primary means by which the government sets out what rules apply to commercial and recreational fishing and other activities regulated under the Fisheries Act 1994. This is necessary for responsible management of fisheries resources, consistent with the main purposes of the Fisheries Act. The Regulation explicitly states that gambusia is a noxious fish. The deliberate release of this fish into Queensland waters is therefore prohibited. A permit is also required under these regulations to release any fisheries resources into Queensland.

	Fisheries (Freshwater) Management Plan 1999	Schedule 5 of the Plan lists those river catchments in which translocations are not permitted. Specifically, translocations are not permitted in all river catchments in the Murray, Lake Eyre, Gulf of Carpentaria and Bulloo-Bancannia drainage divisions, and in 28 river basins in the East Coast drainage division. The stocking of artificially created waters on private land, such as farm dams, is based on Schedule 6 of the Fisheries (Freshwater) Management Plan 1999, which lists fish species available from aquaculture for stocking farm dams and river basins. The basins where each species may be stocked include those in which the fish species occur naturally and those in which the fish species had been stocked regularly as part of the Queensland Government's Recreational Fishing Enhancement Program, resulting in established recreational fisheries prior to the introduction of this legislation. Stocking of farm dams or private waters that meet the requirements of Schedule 6 of the Fisheries (Freshwater) Management Plan 1999 do not require a permit. Approval for other fish species must be sought from DPI	This plan allows the stocking of fish species (including non-indigenous fish) where it has previously been stocked as part of the Queensland Government's Recreational Fishing Enhancement Program and is an established recreational fish. The plan provides clear directions for those responsible for the management of fisheries.

NSW	Fisheries Management Act 1994 and Fisheries Management Amendment Act 2006, which comes into force on 30 June 2008.	The objects of this Act are to conserve, develop and share the fishery resources of the State for the benefit of present and future generations. In particular, the objects of this Act include: (a) to conserve fish stocks and key fish habitats; (b) to conserve threatened species, populations and ecological communities of fish and marine vegetation; (c) to promote ecologically sustainable development, including the conservation of biological diversity; (d) to promote viable commercial fishing and aquaculture industries; a) to promote quality recreational fishing opportunities; b) to appropriately share fisheries resources between the users of those resources; and c) to provide social and economic benefits for the wider community of New South Wales. Part 7A, Division 1 – Threatened Species Conservation provides for the conservation of threatened species and states that one of its objectives is: (a) to eliminate or manage certain processes that threaten the survival or evolutionary development of threatened species, populations and ecological communities of fish and marine vegetation. Part 7, Division 6 provides for the declaration of noxious fish and includes penalties for the sale and possession of noxious fish. It is an offence under Section 211, for a person to either introduce or maintain a noxious fish. Conditions may be included in aquaculture permits for the destruction or control of noxious fish. Section 213 enables the destruction of noxious fish. Part 7, Division 7 of the Act prohibits the release of any live fish (except under a permit) into any waters. However this does not include waterbodies such as farm dams, outdoor ponds or other forms of aquaria. However, this does not include water bodies such as farm dams, outdoor ponds or other forms of aquaria except for class 1 noxious species which cannot be released into private water bodies or kept in aquaria.	Whilst the Act recognises the need to conserve threatened species, it also promotes "quality recreational fishing opportunities". This Act provides for the destruction of noxious fish and aims to eliminate or manage threatening processes. Given the broad scope of these objectives, there could be potential for conflicting management options, in particular, where some species of fish may provide 'quality' recreational opportunities but may pose a threat to conservation. NSW DPI is currently reviewing the NSW noxious fish listings and the associated classes. Changes are anticipated in 12-18 months (pers. comm. Jane Frances)

	Fisheries Management (General) Regulation 2002	The Fisheries Regulation is the primary means by which the government sets out what rules apply to commercial and recreational fishing and other activities regulated under the Fisheries Management Act 1994 and the Fisheries Management Amendment Act 2006. This is necessary for responsible management of fisheries resources, consistent with the main purposes of the Fisheries Act. Part 11 (Protection of Aquatic habitats). Clause 340AA relates to noxious fish and noxious marine vegetation and prohibits the importation of high risk species of fish into NSW. The plague minnow (Gambusia holbrooki) is declared a noxious fish under the Act, but only in waters in a select number of local government areas. Roach tench and redfin perch are specifically excluded from the exemption that applies to aquarium fish (i.e., under section 217 of the Fisheries Management Act)	The Regulations aim to protect aquatic habitats and identify gambusia as a noxious species. Live roach tench and redfin perch are specifically prohibited from being brought into New South Wales.
	Threatened Species Conservation Act 1995 (amended in 2002)	This Act aims to conserve biological diversity, prevent extinction and promote the recovery of listed species populations and ecological communities. The ultimate goal is to recover threatened species, populations and ecological communities, so that their long-term survival in nature can be assured. It provides for the listing of key threatening processes. A threatening process is eligible to be listed if, in the opinion of the Scientific committee (constituted under Part 8 of the Act) it: a) adversely affects 2 or more threatened species, populations or ecological communities, or b) could cause species, populations or ecological communities that are not threatened to become threatened.	The Act lists gambusia as a key threatening process as it is considered to present a serious threat to the survival of threatened species such as the green and golden bell frog, the New England bell frog, and other native frogs.
	The Freshwater Fish Stocking Environmental Impact Statement (EIS) and the NSW Freshwater Fish Stocking Management Strategy (2005)	This strategy outlines the rules, regulations and programs that are designed to manage the activity of fish stocking in the future. Impacts by related activities (such as recreational fishing) or industry sectors (aquaculture and the aquarium trade) are also considered, although the rules applying to such sectors are dealt with under separate management or legislative arrangements. A key priority for the strategy is the introduction of an appropriate management regime to minimise the environmental risks that were identified in the EIS risk assessment. The EIS concluded that many elements of the previous activity of fish stocking posed some threat to ecological sustainability. The strategy recognises that fish stocking can pose significant negative impacts on the environment. Issues such as the translocation of undesirable species and potential impact to threatened species, and the impact of inferior stock on wild populations, are issues which the strategy recognises should be addressed as part of the ongoing management of fish stocking.	These policies do not specifically identify the species of fish of interest in this study. However, the policies could be amended when the strategies are next reviewed, if found necessary at the end of this project.

	Introduction and Translocation Policy 2005	This policy describes controls relating to the introduction and translocation of fish species and fish stocking, including native and non-native species. The policy states: (a) All stockings of fish into NSW water require a permit from NSW Fisheries. (b) NSW Fisheries will not permit any further introductions or translocations of native or non-native species into NSW waters, except as permitted elsewhere in this policy. Penalties apply and are regulated by the Fisheries Management Act 1994.	This policy stipulates that a permit is required to introduce any species of fish into NSW. It therefore reduces the likelihood of pest fish being introduced.

VIC	Fisheries Act 1995 and amendment 2003	The objectives of the Act relevant to this study are as follows: (a) to provide for the management, development and use of Victoria's fisheries, aquaculture industries and associated aquatic biological resources in an efficient, effective and ecologically sustainable manner; (b) to protect and conserve fisheries resources, habitats and ecosystems including the maintenance of aquatic ecological processes and genetic diversity; (c) to promote sustainable commercial fishing and viable aquaculture industries and quality recreational fishing opportunities for the benefit of present and future generations; (d) to facilitate access to fisheries resources for commercial, recreational, traditional and non-consumptive uses; (e) to promote the commercial fishing industry and to facilitate the rationalisation and restructuring of the industry; (f) to encourage the participation of resource users and the community in fisheries management. Part 4, Section 44 of the Act relates to recreational fishing and states that a person must not – (a) take or attempt to take fish from inland water … unless he or she is authorised to do so by a recreational fishery licence. Division 1, Section 69 states that the Governor in Council may, by Order in Council, declare any taxon or community of aquatic flora and fauna to be protected aquatic biota. Division 2, Section 75 states that the Governor in Council may, by Order in Council, declare any aquatic species to be noxious. Division 2, Section 76 states that unless authorized under this Act, a person must not bring into Victoria, or take, hatch, keep, possess, sell, transport, put into any container or release into protected waters any aquatic species that is declared noxious under Section 75. Division 2, Section 85 deals with the seizure and removal of noxious aquatic species and states that an authorized officer may at any time and at any place in Victoria seize and remove any noxious aquatic species from any protected waters. In addition an authorized officer may, by written notice served on any person, require that person to take specified measures to destroy any noxious aquatic species in the possession of that person and to produce evidence of that destruction within a specified period.	The Victorian Act is similar to the NSW Fisheries Management Act in that it highlights the need to protect fish resources and promotes the sustainable management of fishing opportunities; however it also promotes *quality* recreational fishing opportunities, as well as encouraging the facilitation of "rationalising and re-structuring" the industry. As noted above, the desire to maintain quality recreational fishing opportunities could be at odds with the aim to protect native fish. Community participation, particularly the participation of resource users, is also encouraged. Importantly, the Act also empowers the Secretary to "*take any actions necessary*" to delay or prevent the spread of noxious fish species. The Act does not contain any provisions for the introduction of live fish into the State, which is controlled under the Quarantine Act. However, the release of fish into natural water is controlled and it is illegal to release or translocate any noxious species. The stocking of non-indigenous fish is only allowed in future in areas where there is no chance of escape (with the exception of trout). The Act allows the policing of waterways and illegal actions of any person to be controlled.

		Division 2, Section 86 states the Secretary may take any action necessary to delay or prevent the spread of the noxious aquatic species from the water in which it is established to other waters if the Secretary is satisfied that – (a) a noxious aquatic species has become established in any protected waters; (b) there is no practical means of killing or removing the noxious aquatic species. The Act does not specifically define "noxious species".	
	Fisheries Regulations 1998	The commercial fishery license conditions state that any fish or protected aquatic biota taken that are not to be retained (other than noxious aquatic species) shall be immediately returned to the water with the least possible injury or damage. The regulations state that noxious fish cannot be returned to the water alive and it is illegal to hold or translocate the fish.	Gambusia is the only fish of interest in this study identified as a noxious species as declared under the Fisheries Act 1995.
	Flora and Fauna Guarantee Act 1988	The flora and fauna conservation and management objectives are: (a) to guarantee that all taxa of Victoria's flora and fauna other than the taxa listed in the Excluded List can survive, flourish and retain their potential for evolutionary development in the wild; (b) to conserve Victoria's communities of flora and fauna; (c) to manage potentially threatening processes; (d) to ensure that any use of flora or fauna by humans is sustainable; (e) to ensure that the genetic diversity of flora and fauna is maintained; (f) to provide programs— (i) of community education in the conservation of flora and fauna; and (ii) to encourage co-operative management of flora and fauna through, amongst other things, the entering into of land management co-operative agreements under the Conservation, Forests and Lands Act 1987; and (iii) of assisting and giving incentives to people, including landholders, to enable flora and fauna to be conserved; and (f) to encourage the conserving of flora and fauna through co-operative community endeavours. The introduction of live fish into waters is a potentially threatening process and it is an offence "to deliberately or accidentally introduce live fish into public or private waters into a catchment in which the taxon cannot be reliably inferred to have been present prior to 1770AD.	This Act complements the Federal EPBC Act 1999 and the Endangered Species Protection Act 1992 in that it aims to conserve flora and fauna and maintain genetic diversity. It contains provisions to control the spread of new species of fish into catchments where they do not occur.
	Victoria's Fisheries and Aquaculture Research and Development Strategy 2005-2010	Provides a five-year strategy to guide research into commercial and recreational fishing, marine and freshwater aquaculture, and for aquatic ecology and environment.	This strategy recognises gaps in the legislation/literature and identifies the need for additional study into the ecological interaction between exotic species and fishery resources.

SA	Fisheries Management Act 2007	Defines noxious species and exotic aquatic organisms Objects include avoiding, remedying or mitigating adverse effects of activities on the aquatic resources of the State Prohibits introduction of noxious species to the State and their possession/sale etc. except by Ministerial permit Prohibits release of exotic fish into any waters unless permitted Section 83 provides powers to find and destroy exotic aquatic organisms and to limit their spread or impact.	Notes that a lack of scientific certainty re serious threats or irreversible damage should not be used as a reason for postponing measures to prevent damage The yellowfin goby is gazetted as a noxious species in the noxious species list.
	Natural Resources Management Act 2004	Provides for the prevention or control of impacts caused by species of animals or plants that may have adverse effect on the environment, primary production or the community Section 74 provides for the adoption of policies for protection of the environment through the control of pest species of plants and animals Section 75 mandates the preparation of plans for the management of pest species of animals and plants	Redfin perch and gambusia are gazetted as pest species in the pest species list

NT	Northern Territory of Australia Fisheries Act 2005	The key objectives of this Act are as follows: (a) to manage the aquatic resources of the Territory in accordance with the principles of ecologically sustainable development, whether managing a single fish species or an ecosystem, to ensure the promotion of appropriate protection of fish and fish habitats; (b) to maintain a stewardship of aquatic resources that promotes fairness, equity and access to aquatic resources by all stakeholder groups, including (i) indigenous people; (ii) commercial operators and aquaculture farmers; (iii) amateur fishers; and (iv) others with an interest in the aquatic resources of the Territory; and The Act promotes a flexible approach to the management of aquatic resources and their habitats, to promote the optimum utilisation of aquatic resources to the benefit of the community. Division 3, Section 15 relates to releasing fish and/or polluting water without a permit and states that: subject to this Act or to an instrument of a legislative or administrative character made under it a person shall not a) bring into, or release in, the Territory any live aquatic life, live fish, or any live eggs, fry, spat, or larva of fish; b) possess or sell noxious fish or noxious aquatic life; The penalty for any of the above is up to $20,000 or 2 years imprisonment. Division 3, Section 34A relates to the movement of fish and states that: "A person must not move from one place to another place fish or aquatic life of a class prescribed under subsection (4) unless at the time the person moves the fish or aquatic life the person has in his or her possession a document in the approved form specifying the species of fish or aquatic life being moved and the place where the fish or aquatic life was caught, taken or harvested." The Act defines "noxious aquatic life" as "aquatic life that is declared by the Regulations to be noxious aquatic life". The Act defines "noxious fish" as "a fish that is declared by the Regulations to be a noxious fish".	Gambusia is a declared noxious species. There are a number of management plans in place for this particular species of fish, however none exist for the other species of interest in this study. A severe penalty exists under this Act for any illegal importation of any live noxious fish or aquatic life.
	Fisheries Regulations 2006	In accordance with the regulations, a licence is not required for recreational fishing in NT, provided that none of the catch is bartered or sold.	Although a licence is not required to fish in the Northern Territory, penalties are severe for anyone that partakes in illegal activities under the Act.

WA	Biosecurity and Agriculture Management Act 2007	The main purposes of the *Biosecurity and Agriculture Management Act 2007* (BAM Act) are to prevent new animal and plant pests (weeds and vermin) and diseases from entering Western Australia, to manage the impact and limit the spread of those already present in the State, and to safely manage the use of agriculture and veterinary chemicals and ensure agricultural products are not contaminated with chemical residues. Part 2 – Biosecurity This Part of the Act establishes two main points of focus: border biosecurity and biosecurity within the State. This is different from the structure under the existing legislation which separates biosecurity controls by reference to animal pests, plant pests, stock diseases and plant diseases. It is now clearly recognised that the controls should be consistent no matter what type of organism is involved and no matter what agricultural activity or aspect of life or endeavour is being protected in any particular case. Division 1 Permitted, prohibited and unlisted organisms This Division 1 and Division 2 are concerned with border biosecurity – keeping harmful organisms out of the State. Clause 11: permitted organisms The Minister may declare that an organism of a kind specified or described is a permitted organism. As the name implies, “permitted organisms” will, as a general rule, be permitted into W.A. They will be organisms that have been assessed as not posing a biosecurity risk – in other words as not likely to have the adverse effects of “prohibited organisms” (see clause 11). An organism may also be permitted if prohibition on its entry to the State cannot be justified because it is already extensively present in some areas. Section 158 applies to a permitted organism declaration. That section relates to the publication of declarations.	The BAM Act establishes two main points of focus: border security and biosecurity within the State. The BAM Act replaces and integrates 17 existing Acts in the Agriculture portfolio. This includes amendment of the *Fish Resources Management Act 1994* by repealing part 9 of the Act, which deals with noxious fish, and deleting any other references to noxious fish. Under the *Biosecurity and Agriculture Management Act*, noxious fish will be dealt with as declared pests. The BAM Act is 'enabling' legislation. That is, it will enable various things to be done by regulations and other subsidiary regulatory instruments. Much of the operational detail found in the Acts to be replaced will be prescribed by regulations that are to be established under the BAM Act.

		Clause 12: prohibited organisms The Minister may declare an organism of a kind specified or described to be a "prohibited organism". The Minister may do this if there are reasonable grounds for believing that it may have an adverse effect on – • another organism; • human beings; • the environment or part of the environment; or • agricultural, fishing or pearling activities. A declaration of a prohibited organism will usually be made to prevent the entry to W.A. of a harmful organism that is not currently known to be present in the State. However, under clause 12(1)(b), a declaration may also be made in relation to an organism if the organism would have an adverse effect if it were present in the State or a part of the State in greater numbers or to a greater extent. Under clause 15 prohibited organisms may not be imported except in accordance with an import permit and the regulations. Sections 11, 12 and 14 of the BAM Act set out the permitted/prohibited/unlisted organism list system: Section 11: permitted organisms (1) The Minister may declare that an organism of a kind specified or described in the declaration is a permitted organism. (2) Section 157 applies to a declaration made under this section. Section 12: prohibited organisms (1) The Minister may declare that an organism of a kind specified or described in the declaration is a prohibited organism if there are reasonable grounds for believing that the organism — (a) has or may have an adverse effect on — (i)another organism; or (ii) human beings; or (iii) the environment or part of the environment; or (iv)agricultural activities, fishing or pearling activities, or related commercial activities, carried on, or intended to be carried on, in the State or part of the State; or	The Minister may declare an organism to be a 'permitted organism'. Permitted organisms are organisms that may be brought into the State but they must be imported in accordance with any relevant regulations. The Minister may also declare any harmful organism that has, or may have an adverse effect on: any other organism, human being, the environment, or part of the environment, agricultural, pastoral, or other primary industries such as forestry or aquaculture; to be a prohibited organism or declared pest. Declared pests may be assigned to one of three categories. Category 1 – Exclusion: covers declared pests which are not yet present in an area and therefore need to be prevented from entering that area. Category 2 – Eradication: covers declared pests which are present in an area where eradicating them appears feasible. Category 3 – Management: covers declared pests which are present in an area where eradication is not feasible, but where control is necessary. Control could means reducing the numbers, distribution and spread of the declared pests or minimising the harm they do. A person must not keep, breed, cultivate or supply a declared pest (except with an authorisation) and must not release a declared pest into the environment. The same applies to anything infected or infested with a declared pest. Under the BAM Act there is also a duty to report the presence or suspected presence of declared pests and to take measures to control them.

		(b) may have an adverse effect on any of those things if it were present in the State or part of the State, or if it were present in the State or the part in greater numbers or to a greater extent. (2) Section 157 applies to a declaration made under this section. Section 14: unlisted organisms An organism that is not a permitted organism or a declared pest is an unlisted organism. The BAM Act contains various offence provisions with relatively strong penalties. For example: The Act contains a number of further sections detailing the prohibitions, rules and fines as follows: Division 2 — Importing organisms into Western Australia Section 15. Import restrictions Section 16. How to obtain import permit Section 17. Supply of unlawful import Section 18. Possession of unlawful import Division 3 — Biosecurity within Western Australia Section 22. Declared pests Section 23. Dealing with declared pest Section 24. Introducing or supplying declared pest Section 25. Authorised dealing with declared pest Section26. Duty to report declared pest	Management plans can be issued to control a declared pest in an area of the State – which might involve eradication, reduction, prevention of an increase, or prevention of the declared pest harming stock or plants in that area. The BAM Act is currently considered to be the nation's State best-practice legislation that uses a combined permitted/prohibited list system to prevent and control invasive species. The permitted list system controls the potential import of new fish species and ensures all proposed imports of new species are subject to a scientifically robust risk assessment and only those determined low risk permitted for import. The prohibited list system of declared pests controls pests within the State that importantly enables and prescribes State or region-wide prevention and/or eradication action, in addition to traditional control of certain widespread declared pest fish.
TAS	Inland Fisheries Act 1995	Section 128 of this Act states that a person must not remove from, or release into, a registered private fishery any live fish without the written consent of the Director. Part 8 of the Miscellaneous Division states that the Minister, by order, may declare fish to be controlled fish to which this Division applies. Part 150, in relation to controlled fish, states that a person must not: (a) keep or have possession or control of any controlled fish; or (b) consign or convey any controlled fish; or (c) release into any inland waters, water hole, dam, aquarium, hatchery or container any controlled fish. Section 152 states that an Officer may seize and remove from any place any controlled fish.	The Act does not use the term "noxious", rather it refers to a "controlled fish". Gambusia is identified as a "controlled fish". The other species of interest in this study are not controlled fish. It is noted however that the Tasmanian Inland Fisheries Services identifies redfin perch and tench as undesirable species, however there is little statutory control over these species.

	Inland Fisheries (Controlled Fish) Order	Section 4 of the Inland Fisheries Act relates to controlled fish and states that the following are declared to be controlled fish: (a) European carp (*Cyprinus carpio*); (b) freshwater crayfish or yabby of the genus *Cherax*; (ba) Eastern gambusia (*Gambusia holbrooki*); (c) any dead fish of a kind specified in paragraph (a), (b) or (ba); (d) the spawn, fry or young of any fish specified in paragraph (a), (b) or (ba).	The Minister may declare by order for a fish to be controlled under the Inland Fisheries Act. In Tasmania, eastern gambusia has been declared as a fish needing to be controlled.
	Inland Fisheries (Delay or Prevention of Spread of Controlled Fish) Order 2003	Section 3 relates to the delay or prevention of spread of controlled fish. The Director is authorised to take any, or any combination of, the following actions to delay or prevent the spreading of controlled fish to other places or waters: (a) drain inland waters; (b) divert inland waters; (c) manipulate the level of inland waters; (d) augment, restrict, screen or otherwise control inflowing and outflowing waters to inland waters; (e) restrict, totally or partially – (i) the access of any persons or animals to inland waters; and (ii) the activities of persons in or around inland waterways.	As gambusia is listed as a species in need of control, this Order provides for its management in Tasmanian waters.
	Inland Fisheries (Destruction of Controlled Fish) Order 1996	Section 4 relates to the destruction of controlled fish. The Director is authorised to destroy controlled fish in either or both of the following manners: (a) adding any, or any combination, of the following to the inland waters in which the fish are believed to occur: (i) Rotenone; (ii) Alphamethrin; (iii) Endosulfan; (iv) Calcium hydroxide; (b) draining inland waters.	Further to the prevention of spread of Gambusia, this Order allows for its destruction and removal from Tasmanian waterways.
	Management Plans	There are a number of regional management plans for various bodies of water including the Western Lakes - Fishery Management Plan 2002 (TAS).	
MDBC	Murray-Darling Basin Agreement (1992)	This agreement provides for the establishment of the MDB Ministerial Council (decision-making), the MDB Commission (information and management) and an MDB Advisory Committee (consultation and public liaison)	Has no direct management responsibility for the species reviewed. MDBC plays a coordinating role and in this respect is developing a pest fish plan.

9.3 Effectiveness of the legislation

The Australian legislation for dealing with the ecological impacts of alien fish species encompasses three main types of control. The first involves prevention of the importation of potential pest species into Australia based on federal legislation, and into respective states and territories based on state legislation. The second concerns the spread or translocation of fish identified as pests within a state, and the third addresses the mitigation of damage caused by pest fish. Table 9.2 provides a summary of the methods that are or could be used by the various jurisdictions to regulate the spread of the fish species reviewed in this study and to reduce or mitigate impacts caused by them.

Most states (ACT, NT, NSW, VIC, QLD, SA) have some legislation in place under their respective Fisheries Acts that makes it illegal to keep, trade, move, or release live fish into a waterway if the species of fish is declared noxious. NSW currently has three noxious classes: Class 1 species are illegal to keep and trade. Class 2 species can be kept in aquaria. Class 3 species can be kept in aquaria, private garden ponds and farm ponds or dams.(N.B. NSW are currently reviewing these classes and the species lists for them). Tasmania, differs in that pest fish are declared as ‘controlled’ rather than noxious fish, however, this power still comes under its Fisheries Act. Western Australia has adopted a different approach and has a ‘prohibited organisms’ list under its new Biosecurity and Agriculture Management Act 2007. In this respect, management of pest fish has been removed from the umbrella of fisheries management and now rests with biosecurity in Western Australia.

Gambusia is declared as a noxious or controlled species in at least five of the states (NT, NSW, QLD, VIC, TA), and yellowfin goby is listed as noxious in South Australia. The latter listing is surprising given the dearth of evidence of its impact found in this review and its listing can be viewed as a precautionary measure to restrict its spread into South Australia. None of the other species reviewed in this report are listed as either noxious or controlled.

There is legislation in each state to allow a prohibition on the importation, spread or possession of alien pest fish species. In this respect, state legislation is well aligned with Federal legislation which provides for prohibition of import unless a species is listed as suitable for live import into Australia.

Legislation dealing with the management of the impacts of alien pest species within states is not as strongly aligned and coordinated as legislation controlling importation and spread. In New South Wales and Victoria, legislation addressing species conservation allows a fish species to be declared as a ‘key threatening processes’, and this provides a mandate for management. In Victoria, an Action Statement has been produced that outlines a number of provisions for the control of unwanted fish (Department of Sustainability and Environment 2003). New South Wales has listed gambusia as a “Key Threatening Process” and has subsequently produced a comprehensive Threat Abatement Plan for this species (NSW National Parks and Wildlife Service 2003). This approach aligns with similar federal legislation for key threatening processes (Table 9.2), but other states have yet to adopt this approach.

South Australia is similar to New South Wales and Victoria in that its legislation for pest fish management is based on the conservation of natural resources (although primary production is included in this). It too focuses on the management of ‘pest species’ but is somewhat narrower in scope than the broader ‘key threatening processes’ approach. Gambusia and redfin perch are both listed as pest species in South Australia and populations of these fish can be managed through pest management plans.

In contrast to New South Wales, Victoria and South Australia, both the ACT and Western Australia have approached the management of alien pest fish through legislation addressing biosecurity rather than the conservation of natural resources. Gambusia and yellowfin goby are both listed as pest species in the ACT and this listing allows management to restrict their spread as well as to reduce populations.

Management of gambusia within Tasmania is covered by its status as a controlled species and only Queensland and the Northern Territory appear not to have specific legislation for the management of pest fish species present. Management is presumably carried out under the auspices of legislation for the control of noxious fish. Queensland has produced a comprehensive operational strategy for the control of alien freshwater fish species (QDPI 2001).

In general, most states allow the possession of noxious, controlled or pest fish provided a written permit has been obtained. However, there are some potential loopholes in that written permission to release some fish species is only restricted to public waters, thereby allowing the stocking of private waters.

Table 9.2: Jurisdictions and the types of legislative control provided to prevent or manage ecological impacts for the fish species reviewed in this report.

Jurisdictions	Type of control	Species listed	Relevant Acts
ACT	- Noxious fish list and prohibition on possessing and trafficking - Pest species list - Permit required for all live fish imports and release into public waters	- gambusia (pest) - yellowfin goby (pest)	- Fisheries Act 2000 - Pest Plants and Animals Act 2005
NT	- Noxious fish list - Permitted introduction list - Permitted translocations	- gambusia (noxious)	- Fisheries Act 2005
NSW	- Noxious fish list - Key threatening process - Stocking controls	- gambusia (noxious)	- Fisheries Management Act 1994 - Threatened Species Conservation Act 1995 - Fish Stocking Strategy - Introduction and Translocation Policy
QLD	- Noxious fish list - Permitted introductions - Prohibited areas for Introductions	- gambusia (noxious)	- Fisheries Act 1994 - Fisheries (freshwater) Management Plan (1999)
SA	- Noxious species list - Exotic aquatic organisms	- yellowfin goby (noxious) - gambusia (pest) - redfin perch (pest)	- Fisheries Management Act 2007 (noxious species) - Natural Resources Management Act 2004 (pest species)
TA	- Controlled fish declaration	- gambusia (controlled)	- Inland Fisheries Act 1995
VIC	- Noxious fish list - Key threatening process - Fish stocking regulations	- gambusia (noxious)	- Fisheries Act 1995 - Flora & Fauna Guarantee Act 1988
WA	- Prohibited organisms list - Declared pests		- Biosecurity and Agriculture Management Act 2007
MDBC	- Coordination and provision of information for management, including preparation of a pest fish plan	- n/a	- Murray-Darling Basin Agreement (1992).
Commonwealth	- Key threatening process - National live import list - National noxious species list - Invasive species of national importance list	- none	- Environment Protection and Biodiversity Conservation Act 1999

9.4 Summary and conclusions

Coming to terms with federal and state legislation and policies regarding the management of alien fish is complex. This is partly due to the three tiers of government in Australia and therefore the large number of Acts and policies that apply to the control of alien fish. It is also due to an inconsistency in the naming of species between states where the same species is known by several names (e.g., gambusia, mosquito fish, plague minnow). In addition, the various Acts and policies, particularly at a state level, do not provide consistency or clarity in their use of terms such as "non-indigenous," "exotic," and "noxious" fish. Moreover, states have different lists of noxious fish. Where a fish is listed as a noxious species in one state, it is not necessarily listed in another. Such difficulties also apply between states and federal jurisdictions. For example, gambusia, has been declared across different states as "noxious," "a controlled species" and "a threatening process." In New South Wales, it is listed as a Key Threatening Process under the New South Wales Threatened Species Conservation Act 1995, but it was not listed as a Key Threatening Process under the Federal EPBC Act 1991 (Koehn and MacKenzie 2004), presumably because the information provided to meet the Act's criteria was not sufficient to warrant nomination. The criteria and processes for identifying Key Threatening Processes can be expected to differ at both federal and state levels and it might be argued that gambusia is not a significant threat when considered Australia-wide. Given its ability to colonise suitable habitats throughout the continent and its likely widespread impact on a number of native fish and amphibia, this argument would appear to have little credence.

While there may be some differences between the Acts as to how a 'key threatening process' is defined and applied, this review highlights the fact that there are basic inconsistencies and discrepancies between some of the Acts, therefore confusing the treatment of pest fish species. In addition, treatment varies according to external influences on management. For example, the carp control programme in Tasmania was heavily supported by the State Government because control was achievable here. Such assistance was not available to other states where carp are also a problem because the species was so widespread that the problem was too large to address.

River networks are not all contained by state borders, and many fish move freely within these networks. This ecological fact adds to the need for federal and state coordination and consistency to ensure that effective management of pest fish species is achieved at appropriate ecological scales. Koehn and MacKenzie (2004) recommended that a national agency be assigned to coordinate the management of alien pest fish species in order to ensure national consistency and that, as a priority, it adopt a National Alien Fish Management Strategy that recognises roles, responsibilities and cost sharing. We support this recommendation and note the recent national policies that support it such as the Natural Resource Management Standing Committee's commitment to produce a noxious fish list under the ornamental fish strategy in NRMSC (2005), the development of a list of 'pests of national significance' under the Australian Pest Animal Strategy and the adoption of the AusBIOSEC framework that will assist coordination of pest fish management at a national level. In addition, a National Rapid Response Plan for pest fish incursions is currently being developed by ARI for the Invasive Animals CRC.

Coordination is also required within states as well as among them, and there is an emerging trend by various state jurisdictions to provide a more coherent and integrated approach to their biosecurity requirements. Western Australia is leading the way in this direction through its Biosecurity and Agriculture Management 2007 Act, but moves by Queensland and Victoria to consolidate the administration of biosecurity functions may provide the prelude for a review and revision of their respective biosecurity legislation.

A number of mechanisms for threat abatement and pest eradication and control are now available to the various states and, in some states, specific plans for the control of pest fish through threat abatement plans are in place. However, it remains to be seen how effective the operational objectives in these plans will be as the time frames for many are not well specified. This uncertainty over time frames reflects the uncertainty over the feasibility of the various control actions proposed and especially their costs. The implementation and effectiveness of these plans is therefore likely to depend largely on the state finance that is available and allocated to support them, and hence on the level of political support for pest fish species control.

Such resourcing difficulties need to be overcome to make progress and in this respect the AusBIOSEC framework will drive a more integrated national approach to the management of pest alien fish. It has the potential to promote agreed cost sharing agreements to resolve such resourcing difficulties.

10. Summary, conclusions and recommendations

10.1 Summary

The principal aim of this report was to collate and review the information on the environmental impact of six species of alien fish that have established feral populations in Australian waters (Chapter 1). This has been accomplished and both the ecological and genetic impacts of the species have been considered on an Australia-wide basis (Chapters 4 and 5 respectively) and are discussed below.

This assessment necessarily involved a preliminary review of the wide range of environmental impact assessment methods currently used in order to define the strengths and weaknesses of the various approaches. This methodological review (Chapter 3) revealed the complexity involved in unravelling the effects of such alien species on the native fauna from other factors affecting the fauna such as land or water use. The review also revealed the high and often impractical levels of proof required to establish the presence of impacts with scientific certainty. As a consequence, the wide range of ecological studies carried out on two of the species in Australia (e.g., gambusia and redfin perch) were categorised into one of the five progressive stages of impact assessment in order to provide an overview of the cumulative burden of proof. This expresses not only the type of study used and its result, but the overall weight of evidence for impacts. This approach was possible for these two species because of the large number of studies carried out in Australia to date. However, this was not possible for the other species which are much less studied. Hence the assessment of impacts of tench and roach depended largely on the collation of anecdotal information in Australia with knowledge of impacts gained from studies carried out in other countries. This is a more theoretical and hence less robust approach and is more akin to the process of hypothesis generation rather than the provision of proof of impact. Information on the two remaining species (yellowfin and streaked goby) was sparse at both an Australian and international level and so it was not even possible to generate hypothetical models of the impacts of these species. This lack of information meant that it was impossible to assess their potential impact on the Australian environment. Although the approach taken to impact assessment was a species-based one, the synergistic effects of several alien species also need to be considered, especially as gambusia and redfin perch can co-occur, the distribution of tench overlapped that of roach, and both the yellowfin and streaked goby inhabit the inshore marine waters around Sydney and Melbourne.

A review of potential ecological impacts requires knowledge of the species distributions in order to assess the scale of the impacts. New up-dated maps of the current ‘known’ distribution of each species are therefore presented to provide this information (Chapter 2). The maps illustrate both the geographic distribution of each species at the level of catchments occupied, but also indicate the location of individual populations because these are the basic units for managing alien fish. The information used to generate these maps has been stored in Excel files, which can be added to and amended as knowledge of distribution improves, and so provide a template for a national surveillance scheme.

Although some alien species have few redeeming features, others are valued by society for their food or recreational values despite the impacts they may have on the biota and its environment. Therefore, environmental impacts need to be considered within the context of sociological and economic cost-benefits. A sociological and an economic assessment of the potential impacts of the six alien species is therefore included to consider both their societal benefits as well as their potential liabilities (Chapters 6 and 7 respectively). Few studies of this type have been attempted before and it was not surprising that this task proved challenging and suffered from a lack of hard information. Nevertheless, the information that is available was reviewed and whereas the limited hard information is presented, the gaps have also been noted to provide guidance on future socio-economic studies on these species.

This review of impacts was also tasked with over-viewing the current management tools and methods available for the control of these alien species and the policy and legislative environment that directs and constrains their management. An in-depth review of the complex state and federal law surrounding the management of alien fish was beyond the scope of this review, and could be the subject of a book in its own right, nevertheless a summary of the major features of the policy and legislative environment is provided (Chapter 8) together with an account of the main management tools that have been successfully used to control populations of these alien species both in Australia and in other countries (Chapter 9). This information shows that, despite the lack of management tools, useful management is still possible and, in particular, that public education can and needs to play a large role in the management of these fish.

10.2 Conclusions on impacts of gambusia

There is now a weight of evidence provided by a large number of studies in Australia indicating that the primary ecological impact of gambusia is its effect on populations of native fish and amphibia (Chapter 4). No individual study provides irrefutable proof of impact and it is apparent that, in many locations, the impact of gambusia on native fish and amphibia is exacerbated by the impact of human-induced changes in stream habitats. The wide range of environmental factors that can modify the impact of gambusia on native species means that it is difficult to disentangle the effects of individual stressors. This multi-variable nature of the problem is the main impediment to obtaining scientifically defensible proof of impact, and means that proof can only be obtained by an experimental approach that manipulates the abundance of gambusia while the native fauna is monitored to detect change and other variables remain constant. Such field experiments are required for convincing proof of impact and need to be encouraged and supported. Such experiments are likely to be possible at locations where state agencies carry out programmes to control and/or eradicate gambusia and it may be possible to encourage such 'manipulation' studies in conjunction with these control programmes through the provision of targeted funding.

Scientific proof of the impact of gambusia on indigenous biodiversity is likely to be required in the future as management efforts to control gambusia increase in number and size and therefore attract closer public scrutiny of cost and necessity. Even though there are alternative native fish species that can also control mosquito larvae, any increase in human health problems related to mosquitos (e.g., as a consequence of climate change) may result in increased public pressure to use gambusia as a mosquito larval control. Proponents of this will scrutinise the evidence on the impact of gambusia and will need to be convinced of their deleterious effect on the environment. Thus, clear proof of impact will be required to address concerns raised by those people who are concerned about costs of control as well as those people who may wish to spread gambusia for mosquito control. There will be a need for an economic component in such evaluations of cost/benefit and a basis for this has been provided (Chapter 7).

Even though clear and irrefutable proof of the impact of gambusia is currently lacking, the number of independent studies that provide some evidence of an impact on native fish and amphibia is large. This evidence adds to the growing weight of evidence from studies in other countries to indicate that this species can create ecological damage through a reduction in indigenous biodiversity. However, it is also clear that gambusia is not a major problem in some waters, especially those where its densities are kept low (e.g., river reaches subject to large variations in flow). These differing results indicate that the ecological impact of gambusia is modified by a range of environmental factors and can be expected to vary in intensity between locations. At present there is insufficient knowledge of these factors to predict where gambusia will or will not pose a problem, or how much of one.

The wide distribution of gambusia within southern Australia implies that the geographic scope for a reduction in indigenous biodiversity in habitats occupied by this species is potentially large. In this respect, the potential ecological impact of gambusia could surpass that produced by other pest fish species (e.g., common carp), even though it is unlikely to rival that created by the combined effects of land-use changes and water management (e.g., damming, diversion, water abstraction) on aquatic habitats. At present, there is no easy way of comparing the relative impacts of such stressors on aquatic biodiversity except in a qualitative and subjective manner. Common carp are not as widespread as gambusia, but can have a devastating effect on water transparency where they occur. The effect of such a change in water clarity on fish habitats and fish populations is more difficult to gauge because of a lack of studies on the relationships between high turbidity, macrophyte loss and fish habitat. However, it is unlikely that common carp will be solely responsible for localised extinctions of indigenous species of fish and amphibia whereas this is a distinct possibility for gambusia. The impacts of common carp are much more visible than those of gambusia and in this sense gambusia may tend to be overlooked. The fact that biodiversity decline is less obvious to the public eye than water quality decline does not mean that it is ecologically less important. Both affect ecological systems and reduce their resilience and sustainability.

The growing weight of evidence that gambusia does pose problems in many locations in Australia has resulted in the precautionary principle being applied by a number of states and the spread of gambusia is now widely discouraged through public education programmes. Although gambusia has now been spread widely in Australia there are many suitable areas, particularly in the north of Australia where it is not present but where it could be spread to. More widespread and targeted public education about gambusia is therefore needed to counter its spread into such areas. However, there is a danger that management will end here rather than develop proactively to meet future threats. Better information on environmental factors affecting the extent of impacts by gambusia is needed not only to identify new tools for its control but to better predict locations where problems will be greatest and where control will have most effect. In this sense, research will be needed to inform future management so that it develops beyond the public education phase. The Department of the Environment, Water, Heritage and the Arts (DEWHA) will need to facilitate this.

One of the major constraints on management will be the need to develop a national as against a state perspective on gambusia spread and control. For example, it is apparent that gambusia threatens some rare and localised species of indigenous fish and the loss of these would be of national as against state significance. Furthermore, state agencies charged with management do not have the resources to develop tools for gambusia control and, because a lack of tools will hamper management, tool development needs to be accomplished by organisations with a national as against a state focus (e.g., the Invasive Animals CRC). DEWHA therefore needs to support research on alien fish that can be applied across a number of states and therefore has national value. Furthermore, some sort of national coordination in management approaches will be needed because gambusia does not recognise state boundaries. Management of gambusia downstream may be compromised by a lack of management upstream. Finally, there is a need for coordination of a national surveillance system to monitor the status and spread of not just gambusia but other alien pest fish. The DEWHA needs to take the initiative in establishing this through the maintenance of a national database either directly or via another organisation with an Australia-wide focus (e.g., Australian Society for Fish Biology).

10.3 Conclusions on impacts of redfin perch

Redfin perch provide valuable fisheries in some parts of Australia and whereas native fish could also possibly fulfil this role, redfin perch are now well established as a valued recreational fishery in many south eastern and western waters. There are many aquatic environments (e.g., small farm ponds, constructed reservoirs for irrigation and water supply) which are suitable for redfin perch fisheries and this is unlikely to change. However, it is also apparent that the release of redfin perch into many ‘natural’ waters has had a detrimental effect on native fish and some crayfish populations, while the spread of the EHN virus has affected native fish more widely. Although proof of impacts is still circumstantial, the weight of evidence is too great to ignore and, as with gambusia, a precautionary principle needs to be applied to the future spread of redfin perch.

As with gambusia, irrefutable scientific proof of the impacts of redfin perch on native biodiversity is still lacking. Predation is clearly a major mechanism, but this does not exclude competition for food and other mechanisms as these have not been examined in as much detail as in gambusia. As with gambusia, proof of impact will be best achieved via manipulation experiments in the field because a lack of information and/or adequate control sites mean that a replicated BACI approach is unlikely to be feasible.

Whereas there is now widespread acceptance that gambusia can have a detrimental effect on native biodiversity in Australia, the public perception of redfin perch as a pest species is not so strong. This signifies a clear need for better public education. However, this also needs to be informed by scientifically defensible studies of impacts and not by pseudo-scientific ‘spin’ as the latter can undermine genuine attempts to better manage this species. Because the redfin perch is also a valued recreational species, attempts to inform the public of its dangers will need to be coordinated with fishery authorities so that a reasoned and joint approach is created rather than an adversarial one. This requirement emphasises the need for clear scientific proof of impact.

The distribution of redfin perch in Australia is currently limited and falls far short of their potential distribution in southern Australia should their spread not be stopped. A lack of management would therefore greatly increase the potential for large-scale impacts on the native fauna and is not an option. However, the need to balance the current recreational value of redfin perch fisheries with their ecological impacts, means that fishery agencies will need to work with anglers to ensure that their rights are respected. An appropriate common goal would be to identify waters that are and are not suitable for redfin perch fisheries from a solid knowledge of impact mechanisms and location-related effects. The latter is currently lacking so research is needed to inform the future management of this species and to assist with both the potential to develop the fishery (e.g., in closed waters) and to reduce its impact in open waters.

10.4 Conclusions on impacts of tench and roach

These two species are similar in that they are both cyprinids and their current distribution in Australia overlaps (i.e., both are present mainly in Victorian waters, although the tench also occurs in some Tasmanian and southern New South Wales waters).

There is no evidence that these species have created the widespread problems in Australian waters that occur with gambusia and redfin perch and they are currently poorly utilised. However, there is some evidence that tench have contributed to increased turbidity and hence a deterioration in water quality in some, but not all locations where they occur in large numbers. This is likely to be a location-specific effect related to the development of high densities in certain environments. There is therefore a danger that tench could, with other fish species, contribute to the accelerated eutrophication of some waters. Any future stocking or risk of spread needs to keep this possibility in mind.

Neither tench nor roach are piscivorous, nor known to be aggressive to other fish and so any impacts on the native fauna can be expected to be subtle and arise through competition for food, or selective predation on certain species of invertebrate rather than piscivory or displacement from agonistic encounters. Such possibilities have not been examined to date and so remain as hypotheses to be tested.

The highly restricted distribution of roach (confined mainly to Victorian waters, with only one record in New South Wales) and its low fishery value is matched by a lack of data on riverine habitats and river-scale distribution. Roach have a moderate tolerance of salinity so could potentially colonise the lower, tidally-influenced, brackish reaches of rivers which can be bottlenecks for diadromous fish species during their migrations to and from the sea. Better knowledge of this species distribution and abundance is required to assess the scope for impacts and any management requirements. For example, it may be possible to further restrict its distribution (if required) via management of spawning migrations and/or habitats. Conversely, such knowledge will help avert river management actions that may increase its spread or enhance its abundance.

Similar remarks apply to tench and its feeding behaviour. It may have a disproportionate effect on benthic invertebrates, especially Mollusca, and so affect ecosystem functioning by significantly removing these components.

The studies outlined above for roach and tench are not a high priority, but could be readily encouraged through support for post-graduate research via the IACRC or universities. Such studies would help clarify the potential role of these fish in Australian waters and provide a better indication to fish and water managers of their capacity to become pest species either alone or in concert with other alien fish species.

10.5 Conclusions on impacts of yellowfin and streaked gobies

These species are relatively recent arrivals to Australia and unlike the other species reviewed, they inhabit coastal, marine waters and the lower reaches of some rivers. Although they have a relatively isolated and discrete distribution in Australia at present (confined principally to shallow coastal waters around Melbourne and Sydney), it is apparent that they can be spread by ships (via ballast water) and possibly by other means such as coastal currents. The finding of populations of the yellowfin goby in a number of inlets north of Sydney, which are not generally visited by ships, indicates that other vectors can influence their spread. There is therefore an urgent need to survey waters providing optimal habitats for these species north and south of existing infestations in order to monitor their spread and to help identify potential vectors other than ship ballast water. Surveys are also required in other harbours frequented by ships to determine whether these species are more widespread than indicated in the species distribution maps presented in this report.

There is currently no information on the effects of these alien fish on either the coastal ecology of harbours and inlets where they occur, nor the lower regions of rivers in these environments. The shallow coastal environments inhabited by these species are also habitats for a number of native Australian species of goby, and these fish may be displaced by either one of these alien species, or even through a synergistic combination of their effects on the native species. Furthermore, there is scope for inter-breeding between the species of alien and native gobies, which may also lead to a reduction in the native species. Studies targeted at identifying potential impacts are therefore urgently required. Initially these could compare the distribution and abundance of small benthic fish in habitats with and without the alien gobies to determine any difference that maybe attributable to them. Such studies would need to be replicated (e.g., in both Sydney and Melbourne harbours) to ensure that any differences are not isolated and coincidental.

10.6 Recommendations

Lintermans (2004) and Koehn and MacKenzie (2004) have produced a comprehensive series of recommendations aimed at national coordination of the management of alien freshwater fish in Australian freshwaters. The following recommendations amplify and extend these.

1. We recommend that a system is developed for the reporting of fish species occurrence throughout Australia so that a national database can be compiled for both occurrence and absence. This is required for all freshwater fish, not just pest species, and the Australian Government is well placed to promote the development of a set of 'standard' or 'minimum' information requirements for the different sampling methods used so that this key information is recorded and available for future use in the management of fish species. This could be achieved through relevant coordinating groups such as the Pest Fish Working Group (PFWG) and/or the Australian Society of Fish Biologists to develop a reporting system that can be adopted by all states such that the occurrence data are easily imported into state or federal databases.

2. There is a need to ensure that data on fish species occurrence is readily available to all fish managers, and to ensure that databases do not become privatised and/or commercialised to an extent that fishery management and research is compromised. A catchment-wide approach to fish management is required for many fish species, including pest species, so this is especially important where rivers cross state boundaries. This issue has been resolved for the Murray-Darling River network (through the Murray-Darling Basin Agreement) but it also needs to be addressed in other catchments that cross state boundaries. Such catchment-based coordination will be important for the future management of all freshwater fish in Australia, not just pest species, and the Australian Government can play a role in achieving this through the provision of support and guidance to the relevant management agencies.

3. There is an urgent need to ensure that the isolated records of redfin perch and gambusia in river catchments identified in this report are physically checked to determine whether these species still occur at such remote locations and, if so, to determine the desirability and feasibility of their elimination before they spread downstream and damage a much wider area. Populations of gambusia in inland catchments and springs as well as in the Northern Territory are examples of such isolated populations where elimination may be possible as an urgent and high priority.

4. We recommend that a selection of ports, harbours and large estuaries along the eastern and southern coastlines of Australia are surveyed to confirm the absence of alien gobies in these locations and/or to confirm their presence where isolated records have been noted (e.g., Fremantle, Port Kembla). Establishment of this baseline is urgently required in order to monitor the spread of these species. Field studies designed to identify actual or potential impacts of these two fish on the native fauna are also required to determine whether their spread will pose a significant ecological threat or not. These would be best carried out in locations where these sepcies are now abundant (either on their own or together) and

should encompass an initial scoping study followed by a larger study focussed on impact detection. This should also include the Japanese goby (*Tridentiger trigonocephalus*), which is another alien species present in similar locations, but not included in the brief for this review.

5. There is a need to develop a strategy for community (including indigenous community) and stakeholder consultation and education over the management of alien pest fish species at both regional and national scales. Whereas state agencies with responsibilities for pest fish management can be encouraged to carry this out at a state level, a national strategy is required and could be supported by the Australian Government and addressed by agencies with national representation such as the PFWG.

6. We recommend that more public education is provided to restrict the spread of gambusia and redfin perch. For example, information on alternative controls for mosquito larvae in small ponds and water bodies needs to be made more readily available to the public.

7. We recommend that full BACI (before/after control/impact) and/or manipulation-type studies are undertaken for the assessment of impacts of alien fish species on indigenous species. Such impact assessment studies are still required for gambusia and perch despite the weight of evidence for their impact on indigenous fauna. The reason for this is that future management to contain and or control specific populations of such pest fish will come under close public scrutiny because it is likely to be costly and/or the methods proposed may be opposed by some sectors. A major argument used to prevent or delay management of pest fish species is a lack of scientifically defensible evidence of impact. We recommend that governments do all in its powers to ensure that such robust impact studies are carried out by encouraging universities and research agencies (e.g., to carry out the research). In connection with this, we recommend that the Australian Government and key stakeholders identify mutually agreed levels of 'proof of impact' for gambusia and redfin perch respectively as without this, acceptance of impacts and the need for control may be resisted.

8. We recommend that economic baselines on the costs and benefits of gambusia and redfin perch be established as both these species will require increasing management to reduce impacts on indigenous biodiversity. Data on the economic cost/benefit of management will be required to underpin future management strategies.

9. There is no 'one-size-fits-all' approach to the control of alien pest fish species. The development and use of a wide range of tools for pest fish control and impact mitigation needs national support. In particular, there is a need to ensure that the

use of toxicants (such as rotenone) are not prevented because of unfounded fears and/or misinformation about the use of such chemicals. There is also a need to ensure that some research addresses the need for low-tech tools to mitigate impacts (e.g., via habitat manipulation and rehabilitation) and to reduce, as against eradicate, pest fish. Such support can be provided by collating and disseminating information on the use and success/ failure of various pest fish control methods used in Australia and overseas, and by encouraging the development of new tools. Chapter 8 provides an overview of such tools and there are a number of options that are not well developed or utilised because they are as yet untested.

10. Where state agencies carry out a pest fish control exercise, it is necessary to ensure there is support for proper 'before-and-after' studies not just to assess the effectiveness of the control methods on the target species but also to provide tangible evidence of improvements in indigenous species resulting from the control.

11. There is a growing need to coordinate the management of alien pest fish species at a national level through an overarching national management strategy that; (a) identifies priorities at a national level, (b) identifies roles and responsibilities at appropriate geographic scales, and (c) which can provide advocacy of national requirements such as database formation, review and coordination of state legislation in accordance with federal legislation, and targeted publicity to generate better public understanding of the issues and more widespread support for the need to manage these species.

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