Invasive Animals Cooperative Research Centre

Risk assessment models for establishment of exotic vertebrates in Australia and New Zealand

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Australian Government
Department of the Environment, Water, Heritage and the Arts

Australian Government
Bureau of Rural Sciences

Invasive Animals CRC

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Foreword

Exotic vertebrates can establish wild pest populations that prey on livestock and poultry, compete with livestock for food, eat valuable crops and cause land degradation through overgrazing. Exotic vertebrates also prey on and compete with native species for food and other resources, and may directly and indirectly modify ecosystems. They may reduce the range and abundance of native species or even cause them to become extinct. Other harm potentially caused by exotic vertebrates includes spreading diseases and hybridising with native species.

There is a risk that new exotic species could establish as wild pests in Australia. When animals escape or are illegally released they can start new populations in the wild that breed and spread. Once an exotic species is widespread, eradication is virtually impossible. Pre-import screening of exotic vertebrates is recognised as a primary and cost-effective tool to prevent the potential harm caused by exotic vertebrates.

The Bureau of Rural Sciences produced this report for the Invasive Animals Cooperative Research Centre. The report provides information to assist government agencies increase public awareness and assess the risks posed by the import and keeping of exotic species. For example, the Australian Government Department of the Environment, Water, Heritage and the Arts has the agreement of the publisher and contributing authors to republish the information that is relevant to the risk assessment processes for assessing the suitability of exotic animals for live import into Australia. This agreement will facilitate the use of information and tools in this report for scientific-based risk assessment in informing decisions under the *Environment Protection and Biodiversity Conservation Act 1999.*

Prof Tony Peacock

Chief Executive

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Summary

Many exotic species are kept and bred in captivity in Australia and New Zealand as companion or hobby animals, or for their commercial or conservation benefits. Frequent applications are made to import and keep new species. Australian and New Zealand Government authorities support a process of risk assessment and risk management to evaluate and manage any threats that imported exotic species could pose to agriculture, the environment and society. Two main factors considered in these assessments are the risk of a species establishing in the wild, and the risk of it causing harm. Attributes found to increase these risks for exotic vertebrates are described in this report. Models are presented for assessing establishment risk for exotic birds and mammals, reptiles and amphibians, and freshwater fish. Factors affecting pest status are also described here, along with their significance for assessing risks of adverse impact.

Australia is an isolated continent with valuable agricultural industries and a highly diverse native flora and fauna. A suite of exotic species has established wild populations on the mainland: at least 25 mammals, 31 freshwater fish, 20 birds, four reptiles and one amphibian. Additional species have established on Australia's offshore islands.

New Zealand has a similar record, with at least 27 mammals, 17 freshwater fish, 35 birds, one reptile and three amphibians having established exotic populations on the two main islands.

Many of these introduced species are now pests and have adverse impacts on agriculture and the environment. Pre-import screening of exotic vertebrates is recognised as a primary and cost-effective tool to prevent the potential harm caused by exotic vertebrates (Natural Resource Management Ministerial Council 2007).

This project was commissioned by the Invasive Animals Cooperative Research Centre to validate and refine risk assessment models used in decisions to import and maintain exotic vertebrate species. It builds on earlier work conducted by the Bureau of Rural Sciences for the Australian Government Department of the Environment, Water, Heritage and the Arts. The findings will guide future decisions on the import of new species, and on restrictions imposed on exotic species already kept in Australia and New Zealand.

Risk of establishment

Assessing invasion risk relies on identifying factors that are linked to successful establishment if a new species is released. There is considerable scientific literature on the ecological theory of invasions, proposing a suite of factors that may influence whether or not species will establish in new environments.

Factors affecting establishment success have been investigated from data sets for:

- exotic birds and mammals introduced to New Zealand, Australia and the United Kingdom
- exotic reptiles and amphibians introduced to the Britain, California and Florida (United States)
- exotic freshwater fish introduced to ten countries around the world.

Overall, results showed there are four key factors for which there is strong evidence of a correlation with establishment success:

1. Propagule pressure — the release of large numbers of animals at different times and places enhances the chance of successful establishment.
2. Climate match — exotic species have a greater chance of establishing if they are introduced to an area with a climate that closely matches that of their original range.
3. History of establishment elsewhere — a history of previous successful establishment is a strong predictor for all vertebrate taxa.
4. Taxonomic group — species that belong to families and genera that have high establishment success are more likely to be successful than other species, all else being equal.

These results were used to develop and refine models to calculate the establishment risk of exotic species. This report presents updated risk assessment models for the introduction of birds and mammals, of freshwater fish, and of reptiles and amphibians to Australia. The report also includes new models to assess the risk that mammals and birds could establish in New Zealand. Instructions for the use of each model are presented. They are compatible with either the version of CLIMATE that runs in Microsoft Windows or the web-enabled CLIMATCH program (see http://www.brs.gov.au/climatch) currently being developed by Bureau of Rural Sciences.

Using these simple quantitative models a risk of establishment can be calculated, and a species can be ranked at four levels: low, moderate, serious or extreme. While the models may not estimate the probability of establishment success for every species to a high level of accuracy, the low cost of using such models allows large numbers of potential invaders to be screened. Much of the information needed is readily available in the scientific literature. The internet has also made information more accessible, although it should be scrutinised for relevance, currency and accuracy. A precautionary approach is advisable when data are limited; for example when assessing the risk of establishment for species that have little or no history of previous introductions.

Risk of adverse impact

The potential impacts of exotic species can be classified into three main categories: economic, environmental and social. Some of these impacts will be significant, others subtle, and there are also likely to be potential relationships and flow-on effects between these categories.

Unfortunately, reliable knowledge about impacts is sparse for most exotic species, for two main reasons. Firstly, there has been limited research in this area, particularly for fish, reptiles and amphibians. Secondly, introductions of exotic species have often coincided with other changes, such as habitat fragmentation, land degradation, changed land use, changed water and fire regimes, and the introduction of other exotic species.

Decisions about which species are safe to import because they are perceived to pose a low risk of harm will therefore be subject to some uncertainty. There is insufficient reliable knowledge of the factors correlated with impacts of most exotic species to make the development of a quantitative model feasible for assessing the risks of impact for exotic reptiles, amphibians or freshwater fish. However, review of factors associated with adverse impacts indicates that an increased risk is associated with exotic species that:

- have adverse impacts elsewhere
- have close relatives with similar behavioural and ecological strategies that cause adverse impacts elsewhere
- are generalist feeders
- are predatory
- destroy or modify vegetation or otherwise cause major habitat changes
- have the potential to cause physical injury
- harbour or transmit harmful diseases or parasites
- have potential to hybridise with close relatives among native species
- are known to spread rapidly following their release into new environments.

This list could be used as a checklist to make a qualitative assessment of the threat of impacts posed by the establishment of new exotic species in Australia and New Zealand. However, an absence of these factors cannot be taken to indicate that there is a low risk of harm.

Glossary

allopatric	occurring in separate, non-overlapping geographic areas
allotopic	with overlapping ranges but not occurring together
anuran	an amphibian of the order Anura (ie a frog)
conspecific	of or belonging to the same species
detritus	particles of organic material derived from dead or decomposing organisms
detrivore	detritus eater
diurnal	active in daylight hours
extirpate	exterminate
fecundity	average number of females produced by females surviving to reproductive age
gravid	carrying developing young or eggs
heterospecific	of or belonging to a different species
hypoxic	low oxygen levels
intraguild predation	killing and eating among potential competitors
propagule	a number of individuals of a species that could aid in dispersal of that species and from which a new population may establish
propagule pressure	a measure of the number of individuals of a species introduced to an area and the number of discrete release events : the higher the numbers, the greater the pressure
r	intrinsic rate of increase of a species
recipient habitat	habitat into which a new species is introduced
stochasticity	lacking any predictable order or plan; random, unpredictable
substrate	underlying material, for example river bed
sympatric	occupying the same or overlapping geographic areas without interbreeding
syntopic	occurring in the same habitat within the same geographic range
trophic	involving feeding habits or relationship of different organisms in a food chain

1. Introduction

This report brings together reviews and models taken from previous reports prepared for the Vertebrate Pests Committee on exotic birds and mammals introduced to Australia (Bomford 2003) and the then Australian Government Department of the Environment and Heritage (Bomford 2006), including models on exotic freshwater fish introduced to Australia (Bomford and Glover 2004) and exotic reptiles and amphibians (Bomford et al 2005). This report builds on this earlier work and also includes new models developed for exotic mammals and birds introduced to New Zealand and for reptiles and amphibians and freshwater fish introduced to Australia.

Since the earlier models were produced, there have been some changes to the species records. For example, two species of exotic freshwater fish — the pearl cichlid and the rosy barb — that were listed as having failed to establish in Australia by Bomford and Glover (2004) have now established in Australia. Also, the exotic reptile and amphibian database prepared by Kraus (in press) now contains introduction records for many more species than the earlier version used by Bomford et al (2005).

The previously released models presented in Sections 2.5, 3.4, 3.5 and 4.3 of this report have not been altered to take account of these changes to the species records, because these models have already been used to assess many species, and it is desirable that the basis for these assessment be published.

The new models presented in Sections 2.6, 3.6 and 4.4 of this report are based on the more recent species lists used by (Bomford et al 2008 and unpublished data). The reviews of factors affecting establishment success and pest status for exotic reptiles and amphibians (Sections 3.2 and 3.7) and exotic freshwater fish (Sections 4.1 and 4.5) in this report have largely been taken from the previously released reviews for these taxa (Bomford and Glover 2004, Bomford et al 2005) with the inclusion of some more recent literature.

The full data sets used to develop the risk assessment models presented in this report will be available on www. feral.org.au.

1.1 Establishment

Exotic mammals, birds, reptiles, amphibians and freshwater fish have established wild populations in Australia and New Zealand. These exotic species cause significant economic, environmental and social harm (Arthington et al 1999, Kailola 2000, Clarke et al 2001, Bomford and Hart 2002, Bomford 2003, Lever 2003, Long 2003, Kraus in press). There is a pressing need to formulate scientifically sound methods and approaches in the field of risk assessment for exotic species, to minimise the risk that new exotic species do not establish wild populations (Anderson et al 2004). Ecologists continue to suggest and test a large number of factors in search of a set that is consistently associated with establishment success, and risk analysts continue to recommend their use in risk-management schemes (Kolar and Lodge 2002, Bomford 2003, Stohlgren and Schnase 2006, Hayes and Barry 2008). The term 'established' uniformly refers to self-maintaining wild populations of non-native species.

Exotic species are commonly introduced to be kept in captivity for scientific, ornamental or recreational purposes (Bomford 2003). Governments receive applications for the import and keeping of new exotic species, and require guidance on the economic and environmental risks that these species could pose. Hayes and Barry (2008) examine 24 studies that identify correlates of establishment success across six animal groups. They found that only three characteristics were consistently associated with establishment success across taxa: climate/habitat match, establishment success elsewhere, and propagule pressure (number of released individuals and/or number of release events). They conclude that risk managers can place faith in risk assessments based on these factors, while warning that results must be interpreted carefully. Hayes and Barry (2008) also report a suite of biotic factors such as body size, diet, offspring per year, growth rate, lifespan and adaptation to disturbed habitat that have been tested in various studies, but none that have been demonstrated to have consistent effects across taxa. They found an inconsistent association (significant in some studies but not in others) for four species-level characteristics: diet, world geographic range size, non-migratory behaviour and plumage dichromatism.

Models to assess the risk that exotic vertebrate species could establish in Australia have been developed for mammals and birds (Bomford 2003; Bomford et al unpublished data), freshwater fish (Bomford and Glover 2004, Bomford et al unpublished data), and reptiles and amphibians (Bomford et al 2005, 2008).

An exotic species is defined as any species that is introduced to a country that is outside of its natural range. Synonyms for 'exotic' include: 'alien', 'non-native', 'non-indigenous' and 'introduced'. A species can be introduced outside its natural range but still within its country of origin and so be native to that country. Such species are called 'translocated' species. The term 'invasive' has no standard definition, but is generally taken to mean more than just establishment. It usually indicates an exotic species that spreads well beyond its place of introduction and is also often taken to indicate a species that poses a threat to ecosystems, habitats or native species (Richardson et al 2000a, Shine et al 2000). For example, IUCN (2000) states: 'An invasive species means an alien species which becomes established in natural or semi-natural ecosystems or habitat, is an agent of change, and threatens native biological diversity.'

Successfully introduced species differ widely from one another in many attributes including: breeding behaviour, degree of parental care, adult size, feeding habits and preferred habitats. Unfortunately, many of the statistical tests conducted to look for factors that may be correlated with establishment success have lacked power, because sample sizes are often small and because species that have been introduced do not necessarily evenly represent the attributes that ecologists want to test. Hence, a significant effect on establishment success will often only be demonstrated for factors that have a fairly major and consistent effect, such as climate match and introduction effort. Where no significant effect has been found for a factor, such as for diet, migratory behaviour or a tendency to live in disturbed habitats, this does not mean that the factor does not influence establishment success. Expert opinion, published in the scientific literature, suggests that such factors may well be potentially important.

Scientific theory and knowledge are still inadequate for making certain predictions about the invasive capability of individual species. However, predictions of invasion risk by exotic species based on fairly simple risk assessment models (including such factors as climate matching and past invasion success by the species or its close relatives) will allow predictions to be made at low cost to guide management policies and to help inform decisions on import and control. Such simple models may not estimate the probability of establishment success for every species to a high level of accuracy, but the low cost of using such models allow large numbers of potential invaders to be screened. They will also enable government agencies to use their available resources to screen for potential invaders. In contrast, more complicated approaches require intensive, long-term and expensive study, which makes assessments prohibitively expensive and carries no guarantee of improved accuracy. They are unlikely to be an effective use of resources or to deliver outcomes in the timeframes required by applicants or governments for decisions on species suitable for live import.

One problem for creating reliable predictions is the time lag between initial introductions and detectable impact (Ricciardi 2003). Following introduction there is often an initial lag period corresponding to slow population growth and spread that may last years to decades. This may be due to several factors, including density-dependent effects of natural enemies (predators, competitors, diseases and parasites) and genetic selection.

A key component of all the models presented in this report is a species' climate match to Australia and its history of establishing exotic populations elsewhere. The original mammal, bird and fish models were based on the climate-matching program CLIMATE, which was developed for use on Apple Macintosh computers. These models were updated by Bomford (2006) to use the new version of CLIMATE that was adapted by the Bureau of Rural Sciences (BRS) for use on Microsoft Windows PCs (Bureau of Rural Sciences 2006). This new version of CLIMATE is currently being adapted by the BRS to run on the internet as free software called CLIMATCH (Bureau of Rural Sciences; see http://www.brs.gov.au/climatch). The models presented in this report can be used with either the BRS (2006) version of CLIMATE or CLIMATCH. The CLIMATE and CLIMATCH programs both contain data for 16 temperature and rainfall variables (Table 1.1) imported from BIOCLIM (Busby 1991) for 9460 meteorological stations worldwide.

Table 1.1 The 16 climate parameters used to estimate the extent of climatically matched habitat in the CLIMATE/CLIMATCH programs

Temperature parameters (°C)	Rainfall parameters (mm)
Mean annual	Mean annual
Minimum of coolest month	Mean of wettest month
Maximum of warmest month	Mean of driest month
Average range	Mean monthly coefficient of variation
Mean of coolest quarter	Mean of coolest quarter
Mean of warmest quarter	Mean of warmest quarter
Mean of wettest quarter	Mean of wettest quarter
Mean of driest quarter	Mean of driest quarter

This report presents new models to assess the risk that mammals and birds could establish in New Zealand. These New Zealand models are developed for climate matching using either the new version of CLIMATE or the CLIMATCH program. This report also includes the three Bomford (2006) risk assessment models for birds and mammals, for freshwater fish and for reptiles and amphibians, which also use the new CLIMATE or CLIMATCH program.

Risk of establishment is ranked at four levels for all eight models presented in this report: low, moderate, serious or extreme. These risk ranks correspond to the establishment success rates presented in Table 1.2 for the species used to populate the models.

Table 1.2 Establishment success rates for risk assessment models

Establishment Risk Rank	Species success rates[a]
Low	0–6%
Moderate	23–38%
Serious	56–77%
Extreme	82–100%

[a] Percentage of introduced species with this risk ranking that succeeded in establishing an exotic population for the eight models presented in this report.

1.2 Impacts of exotic vertebrates

1.2.1 Types of impact

The potential impacts of exotic vertebrates can be classified into three main categories:

1. Economic impacts including: reduced agricultural productivity or increased production costs, flow-on effects on subsidiary industries, trade effects, damage control costs, decline in property values, and injuries to people or domestic animals.
2. Environmental impacts including: ecosystem destabilisation, reduced biodiversity, reduced or eliminated keystone species, ecosystem destabilisation, loss of habitats, and effects of control measures (an indirect effect). Costs to biodiversity may be difficult or even impossible to quantify.
3. Social and political impacts including: aesthetic damage, impacts on cultural heritage, injuries to people and potential health impacts, reduced quality of life, consumer concerns and political repercussions.

There are likely to be potential relationships and flow-on effects between these categories.

The environmental impacts of exotic vertebrates on an ecological community can be defined as any effect attributable to that exotic that causes, directly or indirectly, changes in the density, distribution, growth characteristics, condition, genetics or behaviour of one or more native populations within that community. This definition is independent of human judgements about the benefits or harm of such impacts.

According to Shine et al (2000), elements for cost assessments for exotic species need to include:

- reduced value of agricultural land
- increased operating costs and loss of income
- damage to buildings and power supplies
- inefficient irrigation
- spread of pests (eg weed seeds) and diseases
- control costs
- loss of sport, game and commercial harvesting
- loss of native species and biodiversity
- ecosystem disturbance and loss and protection, monitoring and recovery costs
- loss of scientific value
- loss of opportunity and ecosystem services for current and future generations
- loss of equitable access to resources.

A very small number of individuals, representing a small fraction of the species' genetic variation in its native range, can be enough to generate massive environmental damage (Shine et al 2000). Ecosystems isolated by geography or evolution, such as those on oceanic islands and in Australia, are often characterised by endemic species and high levels of biological diversity. The evolutionary processes associated with isolation over millions of years make such species especially vulnerable to competitors and predators from other areas (Shine et al 2000). Hence, Shine et al (2000) consider for management purposes that every exotic species needs to be treated as potentially invasive, unless or until there is reasonable indication that it is not so.

They assert that this is why a precautionary approach, based on scientific evidence, should underpin all preventative legal frameworks.

Exotic vertebrates may also have positive impacts. For example, they may be used as biocontrol agents, for research, as companion and hobby animals in the pet trade, as public education and display specimens, or for use in food production (eg edible frogs). However, not all species may be suited to such purposes, or be cost effective to manage once imported.

1.2.2 Demonstrating impact

Although invasive species are widely considered to be a significant threat to biodiversity and agricultural production (Ebenhard 1988, Mack et al 2000), evidence of the ecological impacts of exotic species on native species is frequently absent or anecdotal (Ebenhard 1988; Simberloff 1995, 1997; Vitousek et al 1987, 1997).

According to Hayes et al (2004), there is currently no universally accepted way to measure or estimate the potential impact of non-native species. Indeed this is often the least objective part of any bio-invasion debate because stakeholders (including industry organisations, conservation groups, the hobby sector and governments) may have different values and opinions about what is 'harmful' and what therefore constitutes a negative impact. Harm is most easily defined, and is most easily agreed upon, when it refers to human-health impacts or refers to impacts on certain species, particularly commercially valuable species or endangered ones. Harm is most difficult to define when it refers to potential impacts on species that are of no direct value to people, or to impacts on community structures and ecosystem processes, or where changes to biodiversity or the environment may not be easily measured or may occur over a long period. Identifying species that cause ecological harm is ultimately a subjective process (Hayes and Sliwa 2003). Hutchinson (2001) suggests an absence of hard ecological data on most vertebrates renders the ecological approach unconvincing when trying to predict the behaviour of particular species using ecological models.

A demonstration of environmental impact requires verification of a causal relationship between the presence of an exotic species and changes in a native species' population or a natural community. Rigorous proof of a cause–effect relationship requires an experimental design in which appropriate controls and replications are used. Such experiments have rarely been conducted with the introduction of exotic vertebrates. Less rigorous demonstration of impacts can be obtained by detailed study of a community before and after the introduction of an exotic species. Again, such research is rare because pre-invasion data sets are usually unavailable and because the introduction of exotic vertebrates often occurs concurrently with other changes that make attribution of cause–effect relationships difficult. For some effects however, such as predation on native species by exotic predators, the timing and magnitude of the impact following the introduction make the existence of a causal relationship highly probable. Impact following an exotic introduction may also be demonstrated by experimentally manipulating densities of the exotic species and monitoring community responses.

The best method for developing a predictive model for the impact of vertebrate invasions is to compare the outcomes following multiple introductions of a given species in different ecosystems, to determine if the effects of the invader are consistent and therefore predictable in different environments (Ricciardi and Rasmussen 1998, Ricciardi 2003). Where such multiple introductions of the same species into different communities are associated with similar impacts, this can provide strong inferential evidence of causal impacts. Unfortunately, for most known exotic vertebrates, insufficient quantitative data on impacts are available to make useful comparisons across ecosystems, and the data that do exist are often confounded with impacts due to other factors. Further, there are an increasing number of species being introduced to new environments for the first time that thus have no invasion history from which to draw predictive information.

An alternative approach might be to predict the impact of an introduced species from the invasion history of functionally similar species (Byers et al 2002). It is intuitively appealing to assume that closely related species are functionally similar and will thus have similar impacts. Unfortunately, invasion histories indicate that taxonomic similarity is not a consistent predictor of impact potential (Ricciardi 2003).

According to some ecologists, only about ten percent of exotic species become widespread pests following their establishment (Williamson and Brown 1986; Williamson 1996, 1999; Williamson and Fitter 1996; Enserink 1999; Smith et al 1999). However, a review of the pest status of exotic birds and mammals in Australia and elsewhere, suggests that this generalisation is doubtful for vertebrates and that a more realistic figure for exotic vertebrates is that about half become pests (Bomford and Hart 2002, Bomford 2003). It is not possible to estimate a reliable figure for the percentage of exotic reptiles, amphibians or fish that become pests because few reliable data on the impacts of these taxa are available, particularly for subtle effects such as behavioural and evolutionary changes of native species, habitat and environment changes, food web alterations, and transmission of pathogens. Such effects are rarely investigated (Townsend 1991). Even the negative effects of predation and competition on native species often require long-term, expensive research to demonstrate, and such studies have been conducted for few species. Hence, the absence of evidence of such impacts cannot be interpreted as evidence of absence.

1.3 Assessing risk

The accuracy and consistency of risk assessments, no matter how objective the selection criteria, are largely dependent on the skill and rigor of the assessor. To improve the efficiency and consistency of using risk assessment models, there are opportunities to develop instructive electronic tools to guide operators from different backgrounds (eg operating as the applicant or assessor, or if operating in different jurisdictions).

One problem that can lead to bias is that literature reviews are often restricted to publications in English and global coverage is often neither complete nor uniform across continents (Hayes and Sliwa 2003). Further, even when it is possible to access non-English literature, knowledge about exotic species introductions and their impacts is uneven on a world scale, with more research being undertaken in North America, Australia and Western Europe than elsewhere.

A risk assessment model cannot absolutely determine whether or not an introduced exotic species will establish and if it does what impact it will have (Aquatic Nuisance Species Taskforce 1996). The best that can be achieved is to estimate the likelihood that a species will establish and estimate its potential to cause harm. Likewise, a risk assessment model cannot determine the acceptable risk level (Aquatic Nuisance Species Taskforce 1996). What risk, or how much risk is acceptable, depends on how an agency perceives that risk. Risk levels are value judgments that are characterised by variables beyond the systematic evaluation of information.

There is always uncertainty in risk assessments, and these uncertainties can be divided into three types (Aquatic Nuisance Species Taskforce 1996):

1. Uncertainty of the process (methodology).
2. Uncertainty of the assessor(s) (human error).
3. Uncertainty about the organism (biological and environmental unknowns).

The goal is to reduce the levels of uncertainty as much as possible. Basing the risk assessment methodology on robust scientific knowledge and statistical analyses of past introductions will do much to minimise the first source of uncertainty.

Uncertainty of the assessor(s) is best handled by having appropriately qualified people with an objective approach conducting the assessments. The quality of the risk analysis will, to some extent, always reflect the quality of the individual assessor(s) (Aquatic Nuisance Species Taskforce 1996). Some of the information used in performing a risk assessment is scientifically defensible, some of it is anecdotal or based on experience, and all of it is subject to the filter of perception. Hence, all risk assessments contain a subjective component. Ensuring the assessors have no vested interest in the outcome (leading to a conflict of interest) and that they are appropriately qualified will reduce errors introduced by this second source of uncertainty.

The calibre of a risk assessment is related to the quality of data available, so ensuring that a thorough and comprehensive literature review is undertaken for each species assessed, and that the risk assessment is reviewed by scientists familiar with the species being assessed, can reduce the third source of error.

Species for which little biological data are available represent a potential risk. Although this risk may be small for individual species, the risk becomes much higher if lack of 'demonstrated risk' is used as grounds to import large numbers of species (Aquatic Nuisance Species Taskforce 1996).

It is important that government agencies responsible for live import decisions and management of exotic species take steps to establish and maintain a clear conceptual distinction between assessment of risks, and exotic species management responses or decisions on species suitable for live import. The scientific findings embodied in risk assessments should be explicitly distinguished from the political, economic, and technical considerations that influence the design and choice of policy and regulatory strategies, including any mitigation of identified risks (Aquatic Nuisance Species Taskforce 1996). Hence, risk managers should not attempt to influence the outcome of a risk assessment. Those conducting risk assessments should ensure that they approach the assessment objectively, free from any pressures or motives that might influence the outcome.

2. Exotic mammals and birds

This chapter reviews factors affecting pest status and establishment success of exotic birds and mammals. It provides three models for assessing the risk of establishment and pestiness of birds and mammals. The first two models are for assessing the risk of establishment and the potential pest status of birds and mammals introduced to Australia, updated from Bomford's (2003 and 2006) original model. The third model presented is a new model, for assessing the risk of establishment of birds and mammals introduced to New Zealand. Instructions for the use of each of these models are provided.

2.1 Factors affecting the establishment success of exotic birds

Many factors have been well investigated in studies of birds introduced to New Zealand (Veltman et al 1996, Duncan 1997, Green 1997, Sorci et al 1998, Sol and Lefebvre 2000, Cassey 2001, Forsyth and Duncan 2001, Bomford et al unpublished data) and elsewhere (world — Newsome and Noble 1986, Sol and Lefebvre 2000, Blackburn and Duncan 2001a, Cassey 2002, Cassey et al 2004ab; Hawaii — Moulton and Pimm 1986; St Helena Island — Brooke et al 1995; Australia — Duncan et al 2001; Florida — Allen 2006).

Only two studies have investigated the effect of climate match on establishment success for birds (Duncan et al 2001, Bomford et al unpublished data). Both of these studies found a significant association. Only three studies have investigated the role of introduction success elsewhere as a correlate of introduction success for birds for exotic birds (Brooke et al 1995, Duncan et al 2001, Bomford et al unpublished data): all three studies found a significant association.

Bomford et al (unpublished data) tested four factors for exotic birds introduced to New Zealand for correlations with establishment success: climate match, habitat match, establishment success elsewhere, and propagule pressure (number of release events). This study was a test on independent data of Hayes and Barry's (2008) conclusion that climate match, habitat match and establishment success elsewhere are associated with establishment success across taxa. Bomford et al (unpublished data) also examined two additional factors for birds introduced to New Zealand that Hayes and Barry (2008) found are not consistently associated with establishment success across taxa: migratory behaviour and overseas range size. Birds successfully introduced to Australia have larger overseas range sizes than failed species (Duncan et al 2001). The role of migratory behaviour for birds introduced to New Zealand was unclear, as Veltman et al (1996) and Sol and Lefebvre (2000) found non-migratory behaviour was significantly associated with establishment success, whereas Sorci et al (1998) found this factor was not significant. For birds introduced to Australia, non-migratory behaviour is not significantly associated with establishment success (Duncan et al 2001).

Bomford et al (unpublished data) found the following factors significantly affected establishment success for exotic birds introduced to New Zealand:

- number of release events
- climate match
- establishment success overseas.

Availability of suitable habitat and overseas range size were not significant. Migration appeared to be confounded with other factors and was excluded from Bomford et al's (unpublished data) model. However, Bomford et al found all 16 introduced birds that were obligate migrants in their native range failed to establish in New Zealand and concluded that being an obligate migrant was highly likely to reduce establishment success.

Bomford (2003) discusses a range of additional factors that have been proposed to affect establishment success for exotic birds. Several of these factors have been subjected to quantitative assessment, but only the factors listed above have been demonstrated to show a consistent association with establishment success for birds (Hayes and Barry 2008). Carrete and Tella (2008) found that for pet bird species in Spain, wild-caught birds were highly significantly ($p<0.0001$) more likely to establish wild breeding populations than captive-reared birds, even though captive-reared birds are kept in far higher numbers.

2.2 Factors affecting the establishment success of exotic mammals

Three studies have investigated the role of factors associated with establishment success for exotic mammals (Forsyth and Duncan 2001 — exotic ungulates introduced to New Zealand, Forsyth et al 2004 — exotic mammals introduced to Australia, Bomford et al unpublished data — exotic mammals introduced to New Zealand, Australia and United Kingdom). Forsyth and Duncan (2001) use a sample of only three failed species, so their conclusions may not be robust. Forsyth et al (2004) use modelling to identify factors that significantly contribute to mammal establishment success in Australia. However, their sample size is small relative to the number of factors they test, which may lead to misleading results. Bomford et al (unpublished data) had a larger sample size of introduced mammal data for three countries, but still found that small sample size and confounding between the factors being assessed prevented them from developing a reliable model. Instead, these authors produced and assessed summary statistics to gain insight into the factors influencing establishment success for exotic mammals.

Forsyth et al (2004) and Bomford et al (unpublished data) both found that, relative to failed species, successfully established mammal species:

- had higher average climate matches to the countries where they were introduced
- had larger average world geographic range sizes
- were more likely to have established exotic populations elsewhere
- were introduced more times.

Forsyth et al (2004) also concluded that being non-migratory was marginally significant for successful mammals introduced to Australia. But Bomford et al (unpublished data), working with the larger dataset for three countries, concluded that migratory behaviour appeared unlikely to be important for mammals. Bomford et al (unpublished data) also found availability of suitable habitat was associated with introduction success for mammals introduced to New Zealand. Forsyth and Duncan (2001) found number of released individuals was associated with establishment success.

Bomford (2003) discusses a range of additional factors that have been proposed to affect establishment success for exotic mammals. Forsyth et al (2004) tested the significance of some of these factors for mammals introduced to Australia. Further quantitative assessment is required to determine the role of these factors.

2.3 Risk assessment for the establishment of exotic mammals and birds introduced to Australia

The findings of Duncan et al (2001), Bomford (2003 and 2006) and Forsyth et al (2004) were used to develop models to help guide risk assessments on the likelihood that exotic mammals and birds could establish wild populations if released in Australia. Two models were developed by Bomford (2003 and 2006). The first model has four factors that are strongly linked to establishment risk in the analyses by Duncan et al (2001) and Forsyth et al (2004). The second model includes an additional three factors that many experts suggest are linked to establishment success, but for which there is not such strong quantitative evidence (Bomford 2003 and 2006).

Factors used in Model 1:

1. Degree of climate match between Australia and species' overseas range.
2. Exotic population established overseas.
3. Overseas range size.
4. Taxonomic Class.

Additional factors used in Model 2:

5. Diet.
6. Dwelling in disturbed habitat.
7. Non-migratory behaviour.

Instructions for calculating Establishment Risk Scores from these factors are presented in Section 2.5. A species' score can then be converted to an Establishment Risk Rank of Low, Moderate, Serious or Extreme (Section 2.5). Establishment Risk Scores and Establishment Risk Ranks for exotic mammals and birds introduced to Australia are presented in Appendix A, Table A1.

The numbers of species in each Establishment Risk Rank are presented in Figure 2.1 for the four-factor model and Figure 2.2 for the seven-factor model. These figures both show that the Establishment Risk Ranks for exotic birds and mammals introduced to Australia strongly predict introduction outcomes. Overall, mammals introduced to Australia have a higher establishment success rate (52%) than birds (41%). These success rates are similar for these taxa elsewhere in the world (Bomford 2003).

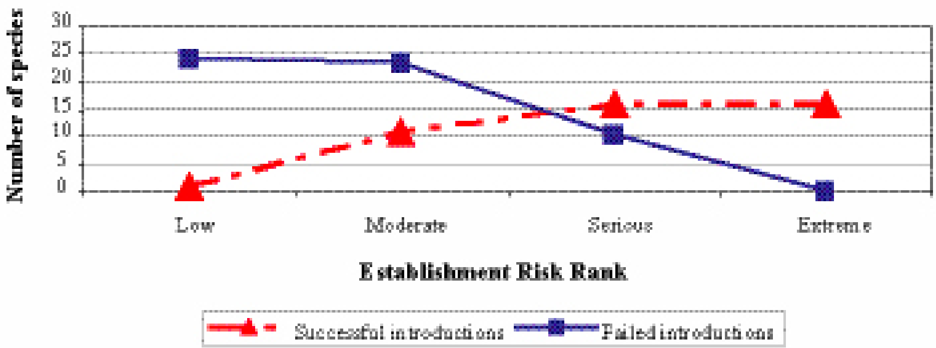

Figure 2.1 Number of species in each Establishment Risk Rank for 101 exotic mammals and birds (combined) introduced to Australia, calculated using four risk factors

Establishment Risk Scores and Ranks were calculated using the directions given for Model 1 in Section 2.5.

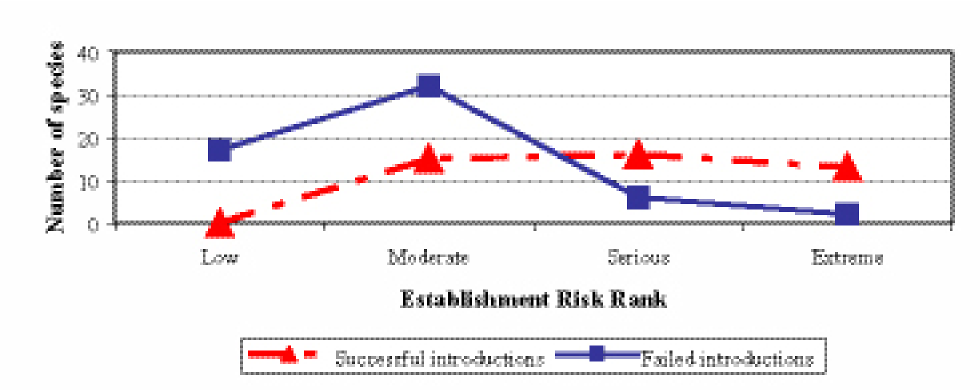

Figure 2.2 Number of species in each Establishment Risk Rank for 101 exotic mammals and birds (combined) introduced to Australia, calculated using seven risk factors

Establishment Risk Scores and Ranks were calculated using the directions given for Model 2 in Section 2.5.

2.4 Risk assessment for the establishment of exotic mammals and birds introduced to New Zealand

The findings of Duncan et al (2001), Forsyth et al (2004) and Bomford et al (unpublished data) were used to develop models to help guide risk assessments on the likelihood that exotic birds and mammals could establish wild populations if released in New Zealand. Each model is the sum of scores for three factors that contribute to establishment risk.

Establishment Risk Scores for mammals are the sum of the following three risk scores:

1. Climate Match Score.
2. Introduction Success Elsewhere Score.
3. Overseas Range Size Score.

Establishment Risk Scores for birds are the sum of these three risk scores:

1. Climate Match Score.
2. Introduction Success Elsewhere Score.
3. Migration Score.

Instructions for calculating these scores are presented in Section 2.6. A species' Establishment Risk Score can be converted to an Establishment Risk Rank ranging from Low to Extreme (Section 2.6). Establishment Risk Scores and Establishment Risk Ranks for exotic species introduced to New Zealand are presented in Appendix B: Table B1 for mammals and Table B2 for birds.

The numbers of species in each Establishment Risk Rank are presented in Figure 2.3 for mammals and Figure 2.4 for birds. These figures both show that the Establishment Risk Ranks for exotic birds and mammals introduced to New Zealand strongly predict introduction outcomes. Overall, mammals introduced to New Zealand have a higher establishment success rate (69%) than birds (33%). These success rates are similar for these taxa elsewhere in the world (Bomford 2003).

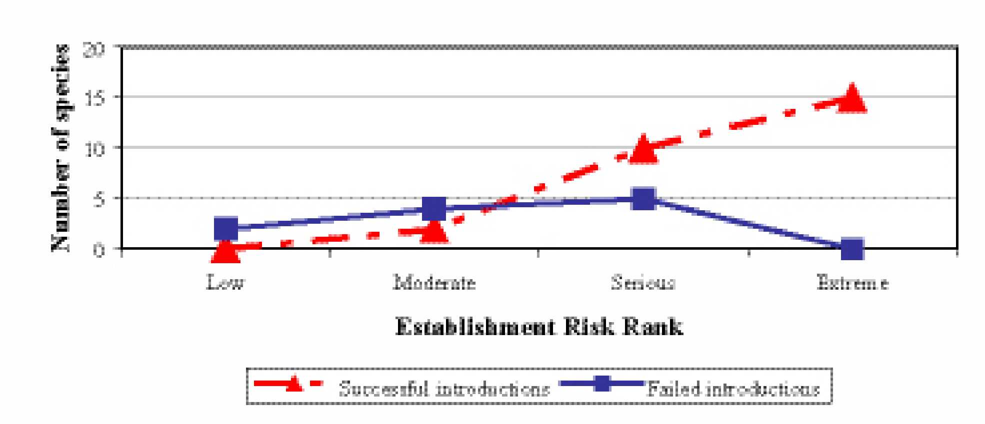

Figure 2.3 Number of species in each Establishment Risk Rank for 38 exotic mammals introduced to New Zealand

Establishment Risk Scores and Ranks were calculated using the directions given in Section 2.6.

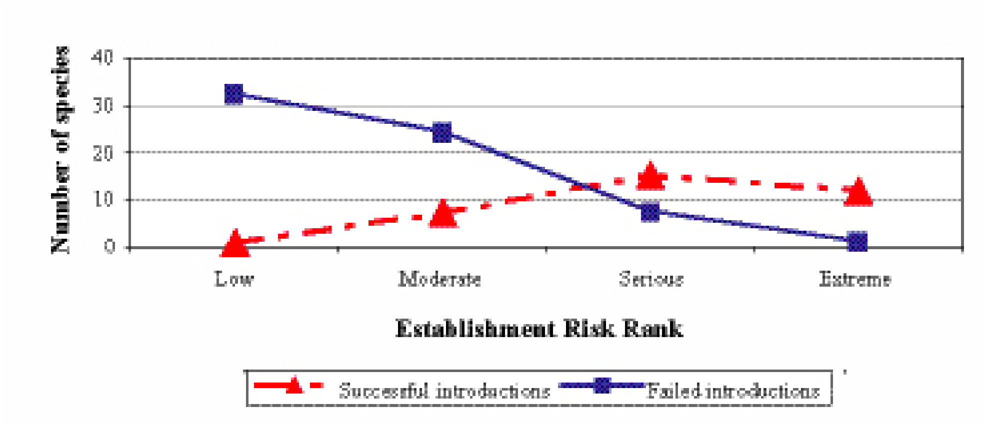

Figure 2.4 Number of species in each Establishment Risk Rank for 99 exotic birds introduced to New Zealand

Establishment Risk Scores and Ranks were calculated using the directions given in Section 2.6.

2.5 Instructions for using the Australian Bird and Mammal Models

The following two risk assessment models for birds and mammals introduced to Australia were developed by Bomford (2003 and 2006). The models apply to the Australian mainland and Tasmania, but NOT to small offshore islands or to marine species. Risk assessments are broken down into four stages: (A) risk posed by captive/ released individuals, (B) risk of establishment, (C) risk of becoming a pest and (D) assigning a Vertebrate Pests Committee (VPC) threat category.

A: Risks to public safety posed by captive or released individuals

A1. Risk to people from individual escapees (0–2)

Assess the risk that individuals of the species could harm people. (NB: this question only relates to aggressive behaviour shown by escaped or released individual animals. Question C11 below addresses the risk of harm from aggressive behaviour if the species establishes a wild population).

Aggressive behaviour, size, plus the possession of organs capable of inflicting harm, such as sharp teeth, claws, spines, a sharp bill, or toxin-delivering apparatus (including toxic skin) may enable individual animals to harm people. Any known history of the species (either captive or wild animals) attacking, injuring or killing people should also be taken into account. Assume the individual is not protecting nest or young. Choose one of the following:

- Animal that sometimes attacks when unprovoked and/or is capable of causing serious injury (requiring hospitalisation) or fatality = 2.
- Animal that can make unprovoked attacks causing moderate injury (requiring medical attention) or severe discomfort but is highly unlikely (few if any records) to cause serious injury (requiring hospitalisation) if unprovoked OR animal that is unlikely to make an unprovoked attack but which can cause serious injury (requiring hospitalisation) or fatality if cornered or handled = 1.
- All other animals posing a lower risk of harm to people (ie animals that will not make unprovoked attacks causing injury requiring medical attention, and which, even if cornered or handled, are unlikely to cause injury requiring hospitalisation) = 0.

A2. Risk to public safety from individual captive animals (0–2)

Assess the risk that irresponsible use of products obtained from captive individuals of the species (such as toxins) pose a public safety risk (excluding the safety of anyone entering the animals' cage/enclosure or otherwise coming within reach of the captive animals):

- Nil or low risk (highly unlikely or not possible) = 0.
- Moderate risk (few records and consequences unlikely to be fatal) = 1.
- High risk (feasible and consequences could be fatal) = 2.

Public Safety Risk Score

A species' Public Safety Risk Score = A = the sum of its scores for A1 and A2.

Public Safety Risk Rank

A species' Public Safety Risk Score is converted to a Public Safety Risk Rank using the following cut-off thresholds:

Public Safety Risk Rank	Risk to Public Safety Score
Not dangerous	A = 0
Moderately dangerous	A = 1
Highly dangerous	A ≥ 2

B. Risk of establishment

The risk of escaped or released individuals establishing a free-living population can be calculated using the four-factor model (Model 1) or the seven-factor model (Model 2) below.

Model 1: Four-factor model for birds and mammals

In this model, an Establishment Risk Rank is calculated using the scores for B1 to B4 below, as outlined.

B1. Climate Match Score (1–6)

Map the selected mammal or bird species' overseas range, including its entire native and exotic (excluding Australia) ranges over the past 1000 years. Use CLIMATE (Bureau of Rural Sciences 2006) or CLIMATCH (Bureau of Rural Sciences; see http://www.brs.gov.au/climatch) and select:

- *'worlddata_all.txt'* as the world data location
- all 16 climatic parameters for matching locations (see Table 1.1)
- Closest Standard Match for the analysis
- Australian splined (gridded) surface for the 'match to' file.

Sum the values for the five highest match classes (ie sum the scores for match classes 10, 9, 8, 7 and 6) = 'Value X'

Convert Value X to a Climate Match Score using the following cut-off thresholds:

CLIMATE Closest Standard Match Sum Level 6 (Value X) (sum of highest five match classes)	Climate Match Score
< 100	1 (Very low)
100–599	2 (Low)
600–899	3 (Moderate)
900–1699	4 (High)
1700–2699	5 (Very high)
≥ 2700	6 (Extreme)

If the input range for a species has 12 or fewer meteorological stations, then it is likely to underestimate the climate match to Australia. If this is the case, it is advisable to increase the Climate Match Score by one increment. For example, if the input range for a species included only five meteorological stations, and the sum of the values for the five highest match classes to Australia equalled 504 (ie Value X = 504), then this would give a Climate Match Score = 2 + 1 = 3.

For domesticated species that originated from wild ancestors more than 1000 years ago, use the feral range of the species where this is applicable. Otherwise an approximate estimate could be obtained either from using the range of the wild ancestor or the range of domestic flocks and herds where they are living in the open with minimal provision of food supplements and shelter.

B2. Exotic Population Established Overseas Score (0–4)

An established exotic population means the introduced species must have bred outside of captivity and must currently maintain a viable free-living population where the animals are not being intentionally fed or sheltered, even though they may be living in a highly disturbed environment with access to non-natural food supplies or shelter. If a species established an exotic population that persisted for at least 20 years before being intentionally eradicated, this can count as an exotic population for the purpose of this question.

This score is calculated as follows:

- No exotic population ever established = 0.
- Exotic populations only established on small islands (< 50 000 km^2; Tasmania is 67 800 km^2) = 2.
- Exotic population established on a larger island (> 50 000 km^2) or anywhere on a continent (including elsewhere on the land mass where the natural distribution of the animal is, if this population is due to human introduction and is geographically separate from the natural range of the species) = 4.

B3. Overseas Range Size Score (0–2)

Estimate the species overseas range size* including currently and the past 1000 years; natural and introduced range in millions of square kilometres.

Overseas range size (million km^2)	Overseas Range Size Score*
<1	0
1–70	1
> 70	2

* A tool for calculating a species' overseas range size will be available with the CLIMATCH program (see http://www.brs.gov.au/climatch).

B4. Taxonomic Class Score (0–1)

This score is calculated as follows:

- Bird = 0.
- Mammal = 1.

Establishment Risk Score

A species' Establishment Risk Score is the sum of its scores for B1 to B4 above.

Establishment Risk Rank

A species' Establishment Risk Score is converted to an Establishment Risk Rank (Low, Moderate, Serious or Extreme) using the following cut-off thresholds:

Establishment Risk Rank	Establishment Risk Score
Low	≤ 5
Moderate	6–8
Serious	9–10
Extreme	11–13

Model 2: Seven-factor model for birds and mammals

In this model, an Establishment Risk Rank is calculated with an additional step to Model 1. First, calculate the scores for B1 to B4 below, as outlined for Model 1:

B1. Climate Match Score (1–6)

B2. Exotic Population Established Overseas Score (0–4)

B3. Overseas Range Size Score (0–2)

B4. Taxonomic Class Score (0–1).

Then, calculate the three additional scores B5 to B7 as described below:

B5. Diet Score (0–1)

This score is calculated as follows:

- Specialist dependent on a restricted range of foods = 0.
- Generalist with a broad diet of many food types or diet unknown = 1.

B6. Habitat Score (0–1)

This score is calculated as follows:

- Requires access to undisturbed (natural) habitats to survive and breed = 0.
- Can survive and breed in human-disturbed habitats (including grazing and agricultural lands, forests that are intensively managed or planted for timber harvesting and/or urban–suburban environments) without access to undisturbed (natural) habitats, or habitat use unknown = 1.

B7. Migratory Score (0–1)

This score is calculated as follows:

- Always migratory in its native range = 0.
- Non-migratory or facultative migrant in its native range, or unknown = 1.

Establishment Risk Score

A species' Establishment Risk Score is the sum of its scores for B1 to B7.

Establishment Risk Rank

A species' Establishment Risk Score is converted to an Establishment Risk Rank (Low, Moderate, Serious or Extreme) using the following cut-off thresholds:

Establishment Risk Rank	Establishment Risk Score
Low	≤ 6
Moderate	7–11
Serious	12–13
Extreme	≥ 14

C: Risk of becoming a pest

C1. Taxonomic group (0–4)

- Mammal in one of the orders that have been demonstrated to have detrimental effects on prey abundance and/or habitat degradation (Carnivora, Artiodactyla, Rodentia, Lagomorpha, Perissodactyla and Marsupialia) = 2.

AND/OR (Score 4 if affirmative for both these points)

- Mammal in one of the families that are particularly prone to cause agricultural damage (Canidae, Mustelidae, Cervidae, Leporidae, Muridae, Bovidae) = 2.
- Bird in one of the taxa that are particularly prone to cause agricultural damage (Psittaciformes, Fringillidae, Ploceidae, Sturnidae, Anatidae and Corvidae) = 2.

AND/OR (Score 3 if affirmative for both these points)

- Bird in one of the families likely to hybridise with native species (including but not restricted to Anatidae, Phasianidae, Cacatuidae and Psittacidae), and if there are relatives in the same genus among Australian native birds = 1.
- Other group = 0.

C2. Overseas range size (0–2)

Estimate the species overseas range size (including current and past 1000 years, natural and introduced range) in millions of square kilometres*:

- Overseas geographic range less than 10 million square kilometres = 0.
- Overseas geographic range 10–30 million square kilometres = 1.
- Overseas geographic range greater than 30 million square kilometres = 2.
- Overseas geographic range unknown = 2.

* A tool for calculating a species' overseas range size will be available with the CLIMATCH program (see http://www.brs.gov.au/climatch).

C3. Diet and feeding (0–3)

- Mammal that is a strict carnivore (eats only animal matter) and arboreal (climbs trees for any reason) = 3.
- Mammal that is a strict carnivore and also strictly ground living = 2.
- Mammal that is a non-strict carnivore (mixed animal–plant matter in diet) = 1.
- Mammal that is a primarily a grazer or browser = 3.
- Other herbivorous mammal or not a mammal = 0.
- Unknown diet = 3.

C4. Competition with native fauna for tree hollows (0–2)

- Can nest or shelter in tree hollows = 2.
- Does not use tree hollows = 0.
- Unknown = 2.

C5. Overseas environmental pest status (0–3)

Has the species been reported to cause declines in abundance of any native species of plant or animal or cause degradation to any natural communities in any country or region of the world?

- Never reported as an environmental pest in any country or region = 0.
- Minor environmental pest in any country or region = 1.
- Moderate environmental pest in any country or region = 2.
- Major environmental pest in any country or region = 3.
- Unknown overseas environmental pest status = 3.

C6. Climate match to areas with susceptible native species or communities (0–5)

Identify any native Australian animal or plant species or communities that could be susceptible to harm by the exotic species if it were to establish a wild population here. Consider specific habitat use and animal behaviour. (For example, if the species being assessed has a score of 1 or more for C3, C4 or C5 above, or for bullets 1 and 4 in C1 above, or if it could compete with, or prey or graze on native species). Compare the geographic distribution of these susceptible plants, animals or communities with the climate match output map of Australia for the species generated by the PC CLIMATE Closest Standard Match analysis (Stage B, Score B1).

- The species has no grid squares within the highest six climate match classes (ie in classes 10, 9, 8, 7, 6, and 5) that overlap the distribution of any susceptible native species or ecological communities = 0.
- The species has no grid squares within the highest four climate match classes (ie in classes 10, 9, 8 and 7) that overlap the distribution of any susceptible native species or communities, and has 1–50 grid squares within the highest six climate match classes that overlap the distribution of any susceptible native species or ecological communities = 1.
- The species has no grid squares within the highest two climate match classes (ie in classes 10 and 9) that overlap the distribution of any susceptible native species or ecological communities, and has 1–9 grid squares within the highest four climate match classes that overlap the distribution of any susceptible native species or ecological communities = 2.
- The species has 1–9 grid squares within the highest two climate match classes, and/or has 10–29 grid squares within the highest four climate match classes, that overlap the distribution of any susceptible native species or ecological communities = 3.
- The species has 10–20 grid squares within the highest two climate match classes, and/or has 30–100 grid squares within the highest four climate match classes, that overlap the distribution of any susceptible native species or ecological communities = 4.
- The species has more than 20 grid squares within the highest two climate match classes, and/or has more than 100 grid squares within the highest four climate match classes, that overlap the distribution of any susceptible native species or ecological communities,

OR

One or more susceptible native species or ecological communities that are listed as vulnerable or endangered under the Australian Government *Environment Protection and Biodiversity Conservation Act 1999* has a restricted geographic range that lies within the mapped area of the highest six climate match classes (ie in classes 10, 9, 8, 7, 6, and 5) for the exotic species being assessed,

OR

Overseas range for the exotic species unknown and climate match to Australia unknown = 5.

List susceptible Australian native species or natural communities that could be threatened.

C7. Overseas primary production pest status (0–3)

Has the species been reported to damage crops or other primary production in any country or region of the world?

- No reports of damage to crops or other primary production in any country or region = 0.
- Minor pest of primary production in any country or region = 1.
- Moderate pest of primary production in any country or region = 2.
- Major pest of primary production in any country or region = 3.
- Unknown overseas primary production pest status = 3.

C8. Climate match to susceptible primary production (0–5)

Assess Potential Commodity Impact Scores (PCIS) for each primary production commodity listed in Table 2.1, based on species' attributes (diet, behaviour, ecology), excluding risk of spreading disease (which is addressed in Question C9), and pest status worldwide as:

- Nil (species does not have attributes to make it capable of damaging this commodity) = 0.
- Low (species has attributes making it capable of damaging this or similar commodities and has had the opportunity but no reports or other evidence that it has caused damage in any country or region = 1.
- Moderate–serious (reports of damage to this or similar commodities exist but damage levels have never been high in any country or region and no major control programs against the species have ever been conducted OR the species has attributes making it capable of damaging this or similar commodities but has not had the opportunity) = 2.
- Extreme (damage occurs at high levels to this or similar commodities and/or major control programs have been conducted against the species in any country or region and the listed commodity would be vulnerable to the type of harm this species can cause) = 3.

Enter these PCIS values in Table 2.1, Column 3.

Calculate the Climate Match to Commodity Score (CMCS) for the species in Australia. Australian Bureau of Statistics (ABS) data for commodity production figures by Statistical Local Area should assist with these assessments. Compare the geographic distribution of susceptible agricultural commodities with the climate match output map of Australia for the species generated by the PC CLIMATE Closest Standard Match analysis (Stage B, Score B1):

- None of the commodity is produced in areas where the species has a climate match within the highest eight climate match classes (ie classes 10, 9, 8, 7, 6, 5, 4 and 3) = 0.
- Less than 10% of the commodity is produced in areas where the species has a climate match within the highest eight climate match classes = 1.
- Less than 10% of the commodity is produced in areas where the species has a climate match within the highest six climate match classes (ie classes 10, 9, 8, 7, 6 and 5) = 2.
- Less than 50% of the commodity is produced in areas where the species has a climate match within the highest six climate match classes AND less than 10% of the commodity is produced in areas where the species has a climate match within the highest three climate match classes (ie classes 10, 9 and 8) = 3.
- Less than 50% of the commodity is produced in areas where the species has a climate match within the highest six climate match classes BUT more than 10% of the commodity is produced in areas where the species has a climate match within the highest three climate match classes = 4.

OR

- More than 50% of the commodity is produced in areas where the species has a climate match within the highest six climate match classes BUT less than 20% of the commodity is produced in areas where the species has a climate match within the highest three climate match classes = 4.
- More than 20% of the commodity is produced in areas where the species has a climate match within the highest three climate match classes OR overseas range unknown and climate match to Australia unknown = 5.

Enter these CMCS values in Table 2.1, Column 4.

Calculate the Potential Commodity Damage Scores (CDS) by multiplying the Commodity Value Indices (CVI) in Table 2.1, Column 2 with the PCIS value in Column 3 and the CMCS value in Column 4, and enter the CDS for each commodity in Column 5. Sum the CDSs in Column 5 to get a Total CDS (TDCS) value for the species, then convert it to a C8 score using the conversion factors given in Table 2.1.

The CVI (in Table 2.1, Column 2) is an index of the value of the annual production value of a commodity. Adjustments to the CVI for a commodity will be required when potential damage by the species is restricted to a particular component of the commodity being assessed. For example, some exotic species may contaminate and consume food at feedlots, and hence cause potential harm to feedlot production of livestock, but not to livestock in the paddock. In such cases, the CVI should be adjusted down in proportion to the value of the susceptible component of the commodity.

Table 2.1. Calculating Total Commodity Damage Score

Column 1	Column 2	Column 3	Column 4	Column 5
Industry	**Commodity Value Index1 (CVI)**	**Potential Commodity Impact Score (PCIS, 0–3)**	**Climate Match to Commodity Score (CMCS, 0–5)**	**Commodity Damage Score (CDS, columns 2 x 3 x 4)**
Cattle (includes dairy and beef)	11			
Timber (includes native and plantation forests)	10			
Cereal grain (includes wheat, barley sorghum etc)	8			
Sheep (includes wool and sheep meat)	5			
Fruit (includes wine grapes)	4			
Vegetables	3			
Poultry and eggs	2			
Aquaculture (includes coastal mariculture)	2			
Oilseeds (includes canola, sunflower etc)	1			
Grain legumes (includes soybeans)	1			
Sugarcane	1			
Cotton	1			
Other crops and horticulture (includes nuts, tobacco and flowers)	1			
Pigs	1			
Other livestock (includes goats, deer, camels, rabbits)	0.5			
Bees (includes honey, beeswax and pollination)	0.5			
Total Commodity Damage Score (TCDS)	–	–	–	

NB The Commodity Value Index scores in this table are derived from Australian Bureau of Statistics 2005–2006 data and will need to be updated if these values change significantly. Directions for completing this table are presented in Stage C, Score C8.

[1]The Commodity Value Index is an index of the value of the annual production value of a commodity. Adjustments to the CVI for a commodity will be required when potential damage by the species is restricted to a particular component of the commodity being assessed. For example, some exotic species may contaminate and consume food at feedlots, and hence cause potential harm to feedlot production of livestock, but not to livestock in the paddock. In such cases, the CVI should be adjusted down in proportion to the value of the susceptible component of the commodity.

Convert total commodity scores to a score for C8 as follows:

TCDS = 0: C8 = 0

TCDS = 1–19: C8 = 1

TCDS = 20–49: C8 = 2

TCDS = 50–99: C8 = 3

TCDS = 100–149: C8 = 4

TCDS ≥ 150 C8 = 5

C9. Spread disease (1–2)

Assess the risk that the species could play a role in the spread of disease or parasites to other animals. This question only relates to the risk of the species assisting in the spread of diseases or parasites already present in Australia. The risk that individual animals of the species could carry exotic diseases or parasites in with them when they are imported into Australia is subject to a separate import risk analysis conducted by Biosecurity Australia.

- All birds and mammals (likely or unknown effect on native species and on livestock and other domestic animals) = 2.
- All amphibians and reptiles (likely or unknown effect on native species, generally unlikely to affect livestock and other domestic animals) = 1.

C10. Harm to property (0–3)

Assess the risk that the species could inflict damage on buildings, vehicles, fences, roads, equipment or ornamental gardens by chewing or burrowing or polluting with droppings or nesting material. Estimate the total annual dollar value of such damage if the exotic species established throughout the area for which it has a climate match of in areas where the species has a climate match within the highest six climate match classes (ie classes 10, 9, 8, 7, 6 and 5, based on the climate match output map of Australia for the species generated by PC CLIMATE Closest Standard Match analysis in Stage B, Score B1).

Convert the property damage risk total annual dollar value to a property damage risk score:

$0	C10 = 0
$1.00–$10 million	C10 = 1
$11–$50 million	C10 = 2
more than $50 million	C10 = 3.

C11. Harm to people (0–5)

Assess the risk that, if a wild population established, the species could cause harm to or annoy people. Consider the risk posed by:

- Species capable of aggressive behaviour, plus the possession of organs capable of inflicting harm, such as sharp teeth, tusks, claws, spines, a sharp bill, horns, antlers or toxin-delivering organs may enable animals to harm people. Any known history of the species attacking, injuring or killing people should also be taken into account (see Stage A, Score A1). Aggressive behaviour reported for wild animals should be given more weight than that reported for captive animals. Take into account aggressive behaviour that may occur when the species is protecting nest or young.
- Non-aggressive species that posses organs or apparatus capable of inflicting harm if handled by people, for example the toxic skin glands on some amphibians.
- Species that can be a social nuisance, especially those that live in close association with people, for example species that invade buildings, or those with communal roosts that can cause unacceptable noise.
- Species that could become a reservoir or vector for endemic parasites or diseases that affect people, the likelihood of transmission to people, and the level of harm caused to people should this occur.

Based on the above assessment, if the species established, score the risk of harm to people as follows:

- Nil risk = 0.
- Very low risk = 1.
- Injuries, harm or annoyance likely to be minor and few people exposed: Low risk = 2.
- Injuries or harm moderate but unlikely to be fatal and few people at risk OR annoyance moderate or severe but few people exposed OR injuries, harm or annoyance minor but many people at risk: Moderate risk = 3.
- Injuries or harm severe or fatal but few people at risk: Serious risk = 4.
- Injuries or harm moderate, severe or fatal and many people at risk: Extreme risk = 5.

Pest Risk Score

A species' Pest Risk Score = C = the sum of its scores for C1–C11.

Pest Risk Rank

A species' Pest Risk Score is converted to a Pest Risk Rank (Low, Moderate, Serious or Extreme) using the following cut-off thresholds:

Pest Risk Rank	Pest Risk Score
Extreme	> 19
Serious	15–19
Moderate	9–14
Low	< 9

D: Decision process — assigning a VPC threat category

To assign the species to a Vertebrate Pests Committee (VPC) Threat category, use the scores from Table 2.2 as the basis for the following decision process.

Risk to public safety posed by captive or released individuals (A= 0–4):

A = 0	Not dangerous
A = 1	Moderately dangerous
A ≥ 2	Highly dangerous

Risk of establishing a wild population

Use Stage B, Model 2 Seven-factor model (B = 1–16):

B ≤ 6	Low establishment risk
B = 7–11	Moderate establishment risk
B = 12–13	Serious establishment risk
B ≥ 14	Extreme establishment risk

Risk of becoming a pest following establishment (C = 1–37):

C < 9	Low pest risk
C = 9–14	Moderate pest risk
C = 15–19	Serious pest risk
C > 19	Extreme pest risk

Table 2.2 Score sheet for Australian Bird and Mammal risk assessment model for assigning VPC threat category

Factor		Score
A1.	Risk to people from individual escapees (0–2)	
A2.	Risk to public safety from individual captive animals (0–2)	
A.	**Risk to public safety from captive or released individuals: A = A1 + A2 (0–4)**	
B1.	Degree of climate match between species overseas range and Australia (1–6)	
B2.	Exotic population established overseas (0–4)	
B3.	Overseas range size (0–2)	
B4.	Taxonomic Class (0–1)	
B5.	Diet (0–1)	
B6.	Habitat (0–1)	
B7.	Migratory behaviour (0–1)	
B.	**Establishment Risk Score: B = B1 + B2 + B3 + B4 + B5 + B6 + B7 (1–16)**	
C1.	Taxonomic group (0–4)	
C2.	Overseas range size (0–2)	
C3.	Diet and feeding (0–3)	
C4.	Competition with native fauna for tree hollows (0–2)	
C5.	Overseas environmental pest status (0–3)	
C6.	Climate match to areas with susceptible native species or communities (0–5)	
C7.	Overseas primary production pest status (0–3)	
C8.	Climate match to susceptible primary production (0–5)	
C9.	Spread disease (1–2)	
C10.	Harm to property (0–3)	
C11.	Harm to people (0–5)	
C.	**Pest Risk Score: C = C1 + C2 + C3 + C4 + C5 + C6 + C7 + C8 + C9 + C10 + C11 (1–37)**	

VPC Threat Category

A species' Vertebrate Pests Committee Threat Category (Natural Resource Management Standing Committee and Vertebrate Pests Committee 2004) is determined from the various combinations of its three risk scores (Table 2.3).

Table 2.3 Vertebrate Pests Committee Threat Categories
Categories are based on: risk posed by captive or released individuals (A), establishment risk (B), and pest risk (C) as described above.

Establish-ment risk[1] (B)	Pest risk[1] (C)	Risk posed by individual escapees (A)	VPC Threat Category
Extreme	Extreme	Highly Dangerous, Moderately Dangerous or Not Dangerous	Extreme
Extreme	High	Highly Dangerous, Moderately Dangerous or Not Dangerous	Extreme
Extreme	Moderate	Highly Dangerous, Moderately Dangerous or Not Dangerous	Extreme
Extreme	Low	Highly Dangerous, Moderately Dangerous or Not Dangerous	Extreme
High	Extreme	Highly Dangerous, Moderately Dangerous or Not Dangerous	Extreme
High	High	Highly Dangerous, Moderately Dangerous or Not Dangerous	Extreme
High	Moderate	Highly Dangerous, Moderately Dangerous or Not Dangerous	Serious
High	Low	Highly Dangerous, Moderately Dangerous or Not Dangerous	Serious
Moderate	Extreme	Highly Dangerous, Moderately Dangerous or Not Dangerous	Extreme
Moderate	High	Highly Dangerous, Moderately Dangerous or Not Dangerous	Serious
Moderate	Moderate	Highly Dangerous	Serious
Moderate	Moderate	Moderately Dangerous or Not Dangerous	Moderate
Moderate	Low	Highly Dangerous	Serious
Moderate	Low	Moderately Dangerous or Not Dangerous	Moderate
Low	Extreme	Highly Dangerous, Moderately Dangerous or Not Dangerous	Serious
Low	High	Highly Dangerous, Moderately Dangerous or Not Dangerous	Serious
Low	Moderate	Highly Dangerous	Serious
Low	Moderate	Moderately Dangerous or Not Dangerous	Moderate
Low	Low	Highly Dangerous	Serious
Low	Low	Moderately Dangerous	Moderate
Low	Low	Not Dangerous	Low

[1]'Establishment Risk' is referred to as the 'Establishment Likelihood' and 'Pest Risk' is referred to as the 'Establishment Consequences' by the Natural Resource Management Standing Committee and Vertebrate Pests Committee (2004).

2.6 Instructions for using the New Zealand Bird and Mammal Models

The following models are for calculating risk of establishment for birds and mammals introduced to New Zealand. The models apply to New Zealand's North and South Islands but NOT to small offshore islands or to marine species.

In these models, an Establishment Risk Rank is calculated using three of the scores for A to D below, depending on whether the species introduced is a bird or mammal, as outlined.

A. Climate Match Score (0–5)

Map the selected mammal or bird species' overseas range — including its entire native and exotic (excluding New Zealand) ranges over the past 1000 years. Use the PC version of CLIMATE (Bureau of Rural Sciences 2006) or the web version of CLIMATCH (Bureau of Rural Sciences; see http://www.brs.gov.au/climatch). Select:

- *'worlddata_all.txt'* as the world data location
- all 16 climatic parameters for matching locations (see Table 1.1)
- Euclidian for the analysis.

Create a New Zealand.clm 'match to' file containing the 70 New Zealand data points from the CLIMATE '*worlddata_all.txt*' dataset (Bureau of Rural Sciences 2006). Create a 'match from'. clm file' incorporating the species' overseas range (excluding New Zealand) and match this to the New Zealand.clm file.

Sum the values for the four highest match classes for each species (ie sum the scores for match classes 10, 9, 8 and 7) = 'Value X'

Convert Value X to a Climate Match Score using the following cut-off thresholds:

CLIMATE Euclidian Sum Level 7 (Value X)	Climate Match Score
0	0
1–40	1
41–50	2
51–57	3
58–59	4
≥60	5

If the input ('match from'.clm file) range for a species has 12 or fewer meteorological stations, then it is likely to underestimate the climate match to New Zealand. If this is the case, it is advisable to increase the climate match score by one increment. For example, if the overseas input range for a species included only five meteorological stations, and the sum of the values for the five highest match classes to New Zealand equalled 6 (ie Value X = 6), then this would give a Climate Match Score of 1 + 1 = 2.

B. Exotic Elsewhere Score (0–4)

This score is calculated as follows:

- Introduced overseas but exotic population failed to establish = 0.
- Introduced but establishment uncertain OR never introduced elsewhere = 2.
- Exotic population established overseas but only on small islands (<50,000 km^2) = 3.
- Exotic population established overseas on larger islands (≥50,000 km^2) or a continent = 4.

C. Overseas Range Size Score for mammals (0–2)

Overseas range sizes are calculated on the breeding range of each species outside New Zealand, including both native and introduced range in millions of square kilometres*.

Overseas range size (million km²)	Overseas Range Size Score
<10	0
10–20	1
>20	2

* A tool for calculating a species' overseas range size will be available with the CLIMATCH program (see http://www.brs.gov.au/climatch).

D. Migration Score for birds (0–2)

This score is calculated as follows:

- Obligatory migrant = 0.
- Non-migratory or partial migratory in bird's native range = 2.

Establishment Risk Score

A species' three risk scores are summed to give an Establishment Risk Score as follows:

- A mammal's Establishment Risk Score is the sum of its scores for A + B + C.
- A bird's Establishment Risk Score is the sum of its scores for A + B + D.

Establishment Risk Rank

The Establishment Risk Score is then are converted to an Establishment Risk Rank ranging from Low to Extreme using the following conversion thresholds

Taxa	Establishment Risk Rank	Establishment Risk Score
Mammals	Low	0–1
	Moderate	2–3
	Serious	4–6
	Extreme	7–11
Birds	Low	0–5
	Moderate	6–7
	Serious	8–9
	Extreme	10–11

2.7 Factors affecting the pest status of exotic mammals and birds

Bomford (2003) reviews the literature on the risk factors for exotic mammals and birds becoming pests and presents a model for quantifying this risk of impact for mammals and birds introduced to Australia.

Bomford (2003) concluded that the following eight attributes may increase risk of adverse impacts, (with the caveat that it cannot be assumed that species without these attributes will not cause harm):

(i) Taxonomic group

Risk of adverse impacts is increased if:

- mammal is in one of the orders that have been demonstrated to have detrimental effects on prey abundance and/or habitat degradation (Carnivora, Artiodactyla, Lagomorpha, Perissodactyla, Rodentia and Marsupialia)
- mammal is in one of the families that are particularly prone to cause agricultural damage (Canidae, Mustelidae, Cervidae, Leporidae, Muridae, Bovidae)
- bird is in one of the taxa that are particularly prone to cause agricultural damage (Psittaciformes, Fringillidae, Ploceidae, Sturnidae, Anatidae and Corvidae)
- bird is in one of the families likely to hybridise with native species (including but not restricted to Anatidae, Phasianidae and Cacatuidae) if there are relatives in the same genus among Australian native birds.

(ii) Overseas range size

If mammals and birds have large overseas ranges (including current and past 1000 years, natural and introduced ranges) they are more likely to establish large geographic ranges where they are introduced, increasing the risk of adverse impacts.

(iii) Diet

If mammals are strict carnivores (particularly arboreal carnivores), or are primarily grazers or browsers, the risk of adverse impacts increases.

(iv) Use of tree hollows

If the species nests or shelters in tree hollows, the risk of adverse impacts increases.

(v) Pest elsewhere

If the species is a pest in its current range, the risk of adverse impacts increases.

(vi) Climate match

If species have a good climate match to areas with susceptible native species or communities, or to areas of susceptible agriculture, fisheries or forestry, the risk of adverse impacts increases.

(vii) Spread disease

If species are capable of assisting in the spread of diseases or parasites already present in Australia, the risk of adverse impacts increases.

(viii) Harm to property

If species are capable of inflicting damage on buildings, vehicles, fences, roads or equipment by chewing or burrowing, or polluting with droppings or nesting material, the risk of adverse impacts increases.

(viii) Harm to people

If species show aggressive behaviour, and possess organs capable of inflicting harm, such as sharp teeth, tusks, claws, spines, a sharp bill, horns, antlers or toxin-delivering organs that may enable animals to harm people, the risk of adverse impacts increases. Any known history of the species attacking, injuring or killing people should also be taken into account. Aggressive behaviour may occur when the species is protecting nest or young.

Some species are a social nuisance, especially those that live in close association with people. Examples are species that invade buildings, or that have communal roosts that cause unacceptable noise. Some species could also become a reservoir or vector for parasites or diseases that affect people. Each of these factors also increases the risk of adverse impacts.

3. Exotic reptiles and amphibians

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This chapter reviews the major pathways for introduction and release of exotic reptiles and amphibians worldwide. It also reviews factors affecting their establishment success. Three alternative models are presented for assessing the risk of establishment of exotic reptiles and amphibians to Australia. The first model is based on single factor analyses by Bomford et al (2005) and Bomford (2006) of reptiles and amphibians introduced to Britain, California and Florida. The second model adapts the Bird and Mammal model described in Chapter 2 of this report for reptiles and amphibians. The third model is adapted from a generalised linear mixed model developed by Bomford et al (2008) on data for reptiles and amphibians introduced to Britain, California and Florida. Instructions for the use of each of these models are provided. Finally, a review of factors that affect the pest status of reptiles and amphibians is presented with implications for risk assessment processes.

3.1 Introduction and release

3.1.1 Reasons for introductions

Kraus (2003) examined published introduction records of exotic reptiles and amphibians around the world. The two major pathways for introductions were intentional movement via the pet trade (34% of introductions) and accidental import in cargo shipments (29%). Introductions via the pet trade involved 72 species, of which 36 species established exotic populations; mainly lizards (37%), turtles (25%) and frogs (22%). Four other pathways also contributed to exotic reptile and amphibian introductions: for human food consumption (9%), for biocontrol (8%), for aesthetic purposes (7%) and accidental introductions associated with the nursery trade (7%).

According to Shine et al (2000), in the modern era of globalisation, the 'four Ts' (trade, transport, travel and tourism) have sharply accelerated the rate of species' movements. The four main reasons given by these authors for exotic species introductions are:

1. Intentional introductions for use in biological production systems (such as agriculture, fisheries, and forestry), and for recreational and ornamental purposes (such as garden ponds).
2. Intentional introductions for use in containments or captivity (zoos, aquaculture, mariculture, aquaria, horticulture, pet trade etc) from which there is a risk of escape or release to the wild.
3. Intentional introductions for biological control of pest species.
4. Unintentional introductions of species through pathways involving transport, trade, travel or tourism.

According to Shine et al (2000), exotic species are routinely introduced to be kept in captivity for scientific, ornamental or recreational purposes. They state 'Once they have been admitted to a new country there is no such thing as zero risk of escape or release.' They further state that 'Deliberate or accidental release of pets and aquarium specimens is a serious problem'.

The desire for novelty leads to a desire for new species to be imported. Some are abandoned out of boredom, carelessness, cost saving, or misguided concern for 'animal welfare'. Internet trafficking in live animals may increase risks.

According to Butterfield et al (1997), introductions in the last 20 years of exotic reptiles and amphibians to Florida are mainly associated with the international pet trade. The rate of introduction of exotic reptiles and amphibians into South Florida was fairly constant from 1940–1958. However, from 1958–1983 the rate of invasion increased three-fold (Wilson and Porras 1983).

The African clawed frog *Xenopus laevis* was shipped around the globe for use in human pregnancy testing during the 1940's and 1950's, leading to exotic populations establishing in parts of Europe, North America, South America, and new areas in Africa (United States Geological Survey 2003a).

Reptiles and amphibians are also frequently imported accidentally in cargo. Hitchhiker or stowaway organisms are inadvertently transported through trade, travel and transport pathways (Shine et al 2000). Such species may breach quarantine barriers. The following are a few of many examples of introductions with cargo:

- Early introductions of exotic reptiles and amphibians to Florida were primarily accidental imports coming in with shipping cargo (Butterfield et al 1997).
- *Eleutherodactylus coqui* and *E. planirostris* frogs were unintentionally introduced to Hawaii via the horticulture trade (Kraus et al 1999, Kraus and Campbell 2002).
- Originally native to the New Guinea area, the brown tree snake *Boiga irregularis* was introduced to Guam, previously a snake-free island, in a shipment of military cargo (United States Geological Survey 2003b).
- The Cuban treefrog *Osteopilus septentrionalis* was first reported in Florida in 1931, and its entry pathway was considered likely to be as a cargo stowaway (United States Geological Survey 2002).
- *Bufo melanostictus* is a large toad widely distributed in Asia but not present in Australia. There have been at least 12 intercepts of *B. melanostictus* at the Cairns port in Queensland (Frank Keenan, pers. comm. 2005). Live individuals have also been detected at least twice at Darwin docks amongst shipments of timber from Malaysia (Tyler 2001).
- The common wolf snake *Lycodon aulicus capucinus* is a recent colonist of Christmas Island in the Indian Ocean. The wolf snake is native to southeast Asia but is not present on the Australian mainland. According to Fritts (1993), it was probably accidentally transported in cargo — such as pallets of timber from Indonesia or the Philippines.

3.1.2 Reasons for release

Animals may be released accidently, or because they are unwanted pets, or because people are intentionally trying to establish wild populations of the species. Examples of exotic reptile and amphibian releases that were found in the literature are listed below.

2361 individuals of 17 species of reptiles and amphibians have been listed as being released in 1964 at the address of an animal dealer in Florida (King and Krakauer 1966).

Nine non-native turtle species were captured from a pond at the University of Davis, California (Spinks et al 2003). With the exception of a marked individual that was stolen from a zoo, the non-native turtles were all species common in the pet and food trades. Spinks et al (2003) suggested that although some introductions may result from the intentional release of 'rescued' individuals intended for human consumption, most of the non-native turtles came from the pet trade. This is because turtles purchased for food must, by law, be slaughtered before sale in California. The majority of turtle species can become quite aggressive and quickly outgrow most aquariums or outlast the owner's commitment to care for them. As a result, some pet owners are unwilling to care for their turtles, and release them into nearby bodies of water. This scenario is particularly likely for the red-eared slider (*Trachemys scripta*), which is the most common turtle in the pet trade (Luiselli et al 1997). Most individuals captured by Spinks et al (2003) were large adults, which are likely to be the most difficult to care for. The immature *T. scripta* captured by Spinks et al (2003) were all hatchlings and yearlings; considered most likely to be offspring of adults released into the university waterway, since juveniles less than 10cm in length are not legally available within the pet trade in the United States.

There have been widespread releases of red-eared sliders in streams and ponds in central Italy by pet keepers who no longer wish to keep them (Luiselli et al 1997). Similarly, according to Cadi et al (2004), there has been 'massive importation' of *T. scripta* as a pet in France over the past few decades, and this has been followed by the release of many of these turtles into natural environments, so the species is now widely distributed in France. According to Cadi and Joly (2004), more than 52 million red-eared sliders were exported from the United States between 1989 and 1997. Many were imported to Europe for private collections, and many were released when they became large and aggressive.

North American bullfrogs (*Rana catesbeiana*) have been widely released throughout the world. The species is prized as food and is also a game species, supporting sport and commercial harvests, although no bullfrog farms have been sustainable. It is also sold for educational and scientific use (Bury and Whelan 1984).

R. catesbeiana larvae have been imported on a large scale to mainland Europe, especially to the Netherlands, Belgium and Germany, and many have been intentionally released as ornamentals in outdoor ponds. This led to establishment of a breeding population in the Netherlands following the release of five tadpoles in 1986 in a newly constructed garden pond (Stumpel 1992).

Scores of exotic free-roaming snakes have been sighted in Hawaii, and these mostly arrived through the smuggling of pet animals (Kraus and Cravalho 2001).

Tiger salamanders (*Ambystoma tigrinum*) have been deliberately introduced as fish bait throughout western United States (Riley et al 2003).

In Florida, release of pets, escape from pet dealers, or intentional release for pest control were common methods of releasing exotic species (King and Krakauer 1966, cited in Wilson and Porras 1983).

Cane toads (*Bufo marinus*) and other species were introduced intentionally as agents of biocontrol around the world (Easteal 1981, United States Geological Survey 2003c). For example, cane toads were introduced into Hawaii from Puerto Rico in 1932 to control sugar cane beetles and other insect pests (McKeown 1978). Similar introductions occurred in Florida, United States Virgin Islands, the Territories of Guam, American Samoa and Australia (McCoid 1995).

3.1.3 Reasons for intentional or assisted spread

There are reports of intentional spread of exotic reptiles and amphibians by people who want to establish new wild populations for hunting, for aesthetic enhancement of gardens and water features, and for biocontrol of pests. Examples of intentional spread found in the literature are listed below.

Spread of *E. coqui* and *E. planirostris* frogs in Hawaii has been rapid, with reported populations increasing from 21 known sites in 1997 to 300 sites in 2001 (Kraus and Campbell 2002). Spread has been largely via the nursery trade, with infested plants sold by retail outlets (including nursery sections in department stores) being a major source of new infestations sites (Kraus et al 1999). Intentional establishment by people has also frequently occurred for two main reasons. Some gardening clubs promoted transporting and releasing frogs in the mistaken belief that these terrestrial frogs would enhance garden ponds and many people moved the frogs around because they liked their calls. Others mistakenly believed the frogs would be a biocontrol agent for pests such as mosquitoes and tropical nut borers *Hypothenemus obscurus* (Kraus and Campbell 2002). Local advocates of *E. coqui* are often unwilling to accept control of the potential pest species, even 'equating invasive-species control with racism' (Kraus and Campbell 2002).

According to McCann (1996), many people in Florida buy *B. marinus* toads and release them in their backyards to control garden insects and slugs. These releases have increased the range of the species and possibly created satellite populations in Palm Beach and Monroe counties. Some people feel that the toads are useful predators and valuable additions to the local fauna.

Hammerson (1982) suggests *R. catesbeiana* bullfrogs may have been accidentally spread in western United States during fish-stocking operations and that people may have also intentionally spread them for hunting.

3.1.4 Control and eradication

The release of a few individuals can lead to a rapidly expanding population that can be difficult to control or eradicate. For example, Campbell and Echternacht (2003) have shown that release propagules of only five individual brown anoles (*Anolis sagrei*) can lead to rapidly expanding populations. So, there may only be a short opportunity to attempt eradication before an exotic population reaches a size where eradication is not feasible.

Eradication of exotic reptiles and amphibians is probably rarely possible because they are so cryptic, usually making it impossible to find all individuals. The exception would be for frogs with obvious calls (like *E. coqui*) that can be used to locate individuals. Even animals as large as pythons can almost certainly not be eradicated despite their size, because they are so cryptic. This makes it more important to ensure that release and establishment of exotic reptiles and amphibians is prevented.

Eradication attempts often fail. For example, feral populations of *Xenopus laevis* have become established in many countries in a relatively diverse range of habitats, and eradication attempts have been unsuccessful (Tinsley and McCoid 1996, Tyler 2001). *X. laevis* is a pest in its native southern Africa, where it spreads through disturbed habitats and interferes with aquaculture. When ponds and rivers dry up during summer drought, *X. laevis* aestivates (sleeps during summer) in underground fissures (Tinsley and McCoid 1996). Poisoning using a range of chemicals, including high concentrations of Rotenone, failed to eradicate the species in California and has even failed to prevent population expansion (Tinsley and McCoid 1996, Lafferty and Page 1997). Such chemical controls are likely to have undesirable effects on native species. While trapping is safer, it is labour intensive and thus expensive. Further, it is unlikely that all adults in a population can be trapped, and this method cannot be used for tadpoles or eggs.

3.2 Factors affecting the establishment success of exotic reptiles and amphibians

3.2.1 Key factors affecting establishment success

Factors affecting establishment success have been investigated for exotic reptiles and amphibians introduced to Florida, California and Britain (Appendix C, Table C1) by Bomford et al (200, 2008) based on analyses of published introduction records collated by Kraus (in press). Four key factors influence establishment success. Relative to failed species, successful species:

- were introduced more times (ie had higher propagule pressure)
- had higher average climate matches to the countries where they were introduced
- were more likely to have established exotic populations elsewhere
- were more likely to belong to a genus or family that had higher success rates elsewhere.

Examples in the literature that support these findings, and implications for risk assessment processes are outlined below.

(i) Number of releases and propagule pressure

Kraus (2003) examined published introduction records of exotic and translocated reptiles and amphibians around the world. He found that the taxa most often introduced were lizards (40% of total introductions) and frogs (30%) followed by snakes (14%), turtles (12%), salamanders (2%) and crocodilians (2%). Kraus (2003) found frogs (76%) and lizards (66%) had the highest establishment success, followed by turtles (56%), snakes (44%), salamanders (33%) and crocodilians (33%). These data, showing that taxa most frequently introduced had the highest introduction success rates, suggest that introduction effort has a strong influence over which species will establish exotic populations.

Wilson and Porras (1983) observed that all exotic amphibians and reptiles that have established in southern Florida because of the pet industry were at some point imported in large numbers and sold at a relatively low price. This suggests that introduction effort probably played a strong role in their establishment success. Many other species that have established exotic populations have also been subject to strong introduction pressure. For example, over 30 million red-eared slider turtles (*T. scripta elegans*) were exported from the United States to 58 countries during 1994–1997, and this contributed to the establishment of this species in temperate and tropical countries around the world (Salzberg 1998, Spinks et al 2003).

For some species, even a small propagule or a single individual may be sufficient to found an exotic population. There are many examples of exotic populations starting on small islands from small introduction propagules. An introduction of seven individuals of *Lacerta sicula* lizards (four females and three males) was sufficient for an exotic population to establish on a small island in the Adriatic Sea — 12 years later, there was a thriving population co-existing with the native *L. melisellensis* population (Nevo et al 1972). Losos et al (1997) introduced populations of *A. sagrei* lizards onto small islands from a nearby source. They introduced propagules of five or ten lizards (2:3 ratio male: female) onto 14 small islands in the Bahamas that did not naturally contain lizards. On all but some of the smaller islands the lizard populations persisted. On some islands the lizards thrived, attaining a population of over 700 individuals on one island. Similarly, Losos and Spiller (1999) released propagules of five individuals (three mostly gravid females and two males) of *A. sagrei* on ten very small islands in the Bahamas. They repeated this experiment on a further ten islands with propagules of five individuals of *A. carolinensis*. The *A. sagrei* populations thrived on nine of the ten islands. In contrast, many of the introduced populations of *A. carolinensis* became extinct. Stumpel (1992) reported successful reproduction of exotic American bullfrogs (*R. catesbeiana*) in the Netherlands. The population started in 1986 from the release of five bullfrog tadpoles into a newly constructed garden pond.

Invasion via multiple loci is the most effective means of establishing exotic species in new environments. For example, *E. coqui* and *E. planirostris* frogs were introduced to Hawaii via the horticulture trade (Kraus and Campbell 2002). Population expansion has been logarithmic and reported populations increased from 21 sites in 1997 to 300 sites in 2001 (Kraus and Campbell 2002).

Risk assessment significance: The number of release events is a significant predictor of establishment success. The total number of individuals released, and the number of sites at which releases occur may also affect establishment success. These three variables, which collectively determine the level of propagule pressure, should be considered as key factors when managing the risk of exotic species establishing in Australia.

The number of reptiles and amphibians that escape or are released is likely to increase if more species are kept, in higher numbers, and in more locations. Hence, propagule pressure can be reduced by restricting:

- which species are kept in Australia
- the number of collections holding a species
- the number of individuals held in each collection
- the security conditions for keeping species.

Educating people about the risks of releasing exotic reptiles and amphibians is also important. Any changes to policy or management for exotic species that allow more species to be imported, or reduce restrictions on where exotic species can be held or the numbers held, are likely to increase the risk that more exotic reptile and amphibians species will establish wild populations in Australia.

(ii) Climate match

Climate match is a measure of the climate similarity between the sites of origin and release, based on rainfall and temperature data. Bomford et al (2005, 2008) found climate match is a significant predictor of introduction success for reptiles and amphibians introduced to Britain, California and Florida. Climate match is also a significant predictor for exotic reptiles and amphibians introduced to other states in the United States (Figure 3.1).

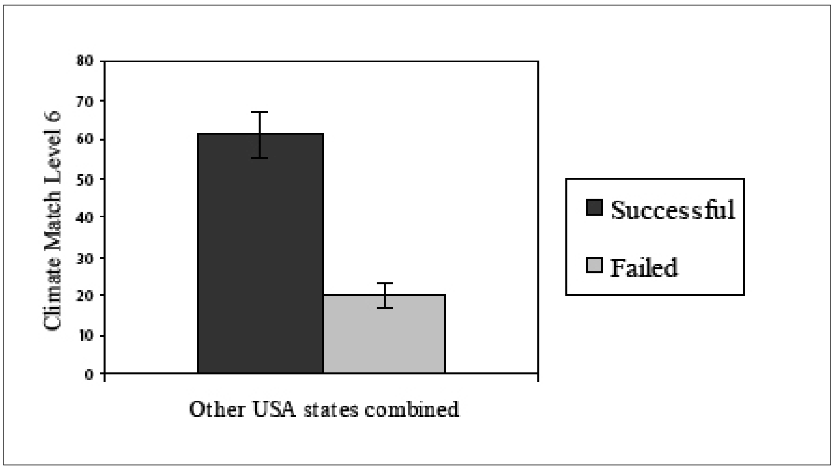

Figure 3.1 Average CLIMATE matches for 41 exotic reptile and amphibian species introduced to 31 United States mainland states (excluding Florida, California, Alaska and Hawaii)
Climate matches are Sum Climate 6 scores expressed as a percentage of the total number of meteorological stations in each state with standard error bars. Scores were calculated for species introductions to each individual state and then the resulting scores were combined into a single data set for all the states (n = 92 introduction records). Species introduction data collated from literature sources by Kraus (in press).

Freezing weather can eliminate newly released propagules of reptiles and amphibians if they are introduced to an inhospitable climate, preventing breeding populations from establishing permanent populations. Wilson and Porras (1983) consider that low temperatures to the north will probably limit the dispersal of many exotic reptiles and amphibians in southern Florida. For example, the original Florida population of the Puerto Rican coqui (*E. coqui*) was eradicated by freezing weather (Wilson and Porras 1983). Wilson and Porras (1983) also suggest freezing weather can exterminate populations of newly established exotic frogs in southern Florida.

Guisan and Hofer (2003) looked at distributions of reptiles in Switzerland and used generalised linear modelling to predict geographic ranges. They found climate, (principally temperature-related factors) accounted for up to 65% (range 6–65%) of deviance, whereas topography (eg altitude, slope and aspect) explained up to 50% (range 0–50%). Low values for both factors were obtained for three widely distributed species: *Anguis fragilis* (slow worm — a limbless lizard), *Coronella austriaca* (smooth snake) and *Natrix natrix* (grass snake).

There has been widespread release of red-eared sliders (*T. scripta*) in streams and ponds in Italy. Luiselli et al (1997) found there were few juvenile *T. scripta* present in the wild in Italy, so they tested an outdoor enclosed population to see if it would breed. No eggs were produced and Luiselli et al (1997) concluded that *T. scripta* introduced to central Italy may have very low, if any, reproductive potential. Luiselli et al (1997) also found juvenile *T. scripta* in an outdoor enclosed population had high winter mortality. In contrast, an enclosed population of the native turtle *Emys orbicularis* both produced eggs and had good winter survival of juveniles. Adult survival of both species was high. Luiselli et al (1997) point out that in its native range *T. scripta* occurs in some very cold areas, and suggests that if individuals from these areas were introduced they might be more successful. Da Silva and Blasco (1995) consider it likely that the similarity of climate and habitat to its native range will contribute to *T. scripta* establishing breeding populations in southwestern Spain.

X. laevis frogs are principally confined to aquatic habitats though able to move overland between water bodies (Measey 1998a; Lobos and Jaksic 2005). Adult *X. laevis* have a wide temperature tolerance, a short generation time (eight months under optimum conditions) and an extended breeding season. In California, breeding is opportunistic, triggered by warm water temperatures and *X. laevis* start breeding at a young age when they are still growing rapidly. Measey and Tinsley (1998) found that exotic *X. laevis* in South Wales (Britain) are only able to breed well enough to achieve major population recruitment about every five years, because wet summers are usually too cool and dry summers too warm. Suitable wet, warm summers are uncommon in Wales and tadpoles may fail to metamorphose before winter (Tinsley and McCoid 1996). Body growth is highly seasonal, limited to warmer months, and occurs at one-third the rate of *X. laevis* in California (Measey 1998a). This unsuitable climate is contributing to a population decline and may help explain why *X. laevis* is not yet a threatening invader in South Wales (Tinsley and McCoid 1996, Measey 1998a).

Climate change may affect the potential ranges for exotic reptiles and amphibians. For example, after some 150 years of relatively unsuccessful introductions of the edible frog *Rana esculenta* into United Kingdom, there is evidence that the species has, within the past decade, suddenly begun to expand its range in the country. Beebee (1995) suggested the species is responding to climate change by altering its breeding cycle times, because populations have spawned progressively earlier over this period, with an overall difference of nearly three weeks.

Risk assessment significance: The level of climate match should be considered as a key factor when assessing the risk that new exotic species could establish. However, climatic match alone is not sufficient to ensure an exotic reptile or amphibian will be able to survive and reproduce. Climatic matching only sets the broad parameters for determining if an area is suitable for an exotic reptile or amphibian to establish. Many factors, such as unsuitable habitat, the absence of suitable spawning habitats or food, or the presence of competitors, predators or diseases, could prevent an exotic reptile or amphibian from establishing in a climatically matched area. Thus, climate matching would usually overestimate the area of suitable climate in the country where a species was introduced. On the other hand, these same biotic and non-climate related abiotic factors could prevent a species from spreading to surrounding areas with suitable climate from its native or current introduced range (Taylor et al 1984). In such a case, climate matching could underestimate the area of suitable climate in the country where a species was introduced.

(iii) History of establishing exotic populations elsewhere

A proven history of invasiveness may indicate that a species has attributes that increase the risk of it becoming a successful invader in other areas (Bomford 1991). Exotic reptiles and amphibians that have a history of establishing exotic populations elsewhere are more likely to establish exotic populations when they are introduced (Bomford et al 2005, 2008).

Risk assessment significance: Because a history of establishing exotic populations elsewhere is a significant predictor of establishment success for exotic reptiles and amphibians introduced to Britain, Florida and California (Bomford et al 2005, 2008), this variable should be considered as a key factor when assessing the risk that exotic reptiles and amphibians could establish in other countries. However, many species that are potential exotics have not been transported to and released in new environments, so they have not had the opportunity to demonstrate their establishment potential. Hence, a precautionary approach is advisable when assessing the risk of establishment in Australia for species that have little or no history of previous introductions.

(iv) Taxonomic group

Exotic species from genera or families with high establishment success are more likely to successfully establish than species from genera or families with low establishment success (Bomford et al 2005, 2008).

Risk assessment significance: For reptile and amphibian species with a history of introductions to new areas, or with close relatives (confamilial) having such a history, previous establishment success rates should be considered a key predictor of future establishment success. A precautionary approach to their introduction is advisable for reptiles and amphibians that have little or no introduction history, and without relatives with an introduction history.

3.2.2 Other factors potentially affecting establishment success

For most of the species they tested, Bomford et al (2005, 2008) were unable to obtain sufficient reliable data on species-level factors (such as diet, offspring per year, growth rate, body size, lifespan or adaptation to disturbed habitat) that could be adapted to a consistent format suitable for making comparisons between species. For example, literature and database searches found even such basic data as body size may be presented as body length or body mass. Even where available, body length may be presented as snout–vent or full-body length and data may be given as average or maximum length. Sex, age, sample size, and captive versus wild status are also often unspecified or inconsistent between species. Data from captive animals fed regularly and maintained at optimal temperatures are not likely to exhibit traits similar to free-ranging individuals (Reed 2005). The situation was worse when trying to obtain reliable ecological data because most herpetological species have yet to be studied.

Although the lack or inconsistent quality of data prevented Bomford et al (2005, 2008) from testing species-level factors in their establishment models, Hayes and Barry (2008) found no species-level factors to be consistently associated with establishment success in other vertebrates or invertebrates. So, it may not be important that Bomford et al (2005, 2008) were unable to test the importance of such factors for reptile and amphibian establishment success, or to include them in their risk assessment model.

The literature suggests there are ten additional factors that may influence establishment success, but for which supporting data are lacking. These factors and implications for risk assessments are outlined below.

(i) Overseas geographic range size

Campbell and Echternacht (2003) suggested the extensive native range of the brown anole *A. sagrei* is one of the characteristics that contributes to its successful invasion history. Bomford et al (2005, 2008) found world geographic range size was not significantly correlated with establishment success for reptiles and amphibians introduced to Britain, California and Florida.

Risk assessment significance: It is doubtful if overseas geographic range size influences introduction success for exotic reptiles and amphibians. Therefore this factor should probably not be taken into account when assessing the risk that exotic reptile and amphibian species could establish in Australia.

(ii) Ability to live in disturbed habitats and human commensalism

Many ecologists consider that an ability to live in human-modified or other disturbed habitats, particularly agricultural or urban/suburban areas, is a major factor contributing to the establishment success of exotic animals (Shine et al 2000, Bomford 2003). Many of the exotic reptiles and amphibians that have established in Hawaii, Florida, California and United Kingdom are able to live commensally with people and usually initially establish in human-disturbed areas. This may, however, be due at least in part to the fact that more releases occur in human-occupied habitats. Further, the slow dispersal abilities of many reptiles and amphibians may not have allowed them to reach native habitats in the relatively short times since their introductions. Exotic reptiles and amphibians in Florida are strongly associated with disturbed areas altered primarily through urbanisation or agriculture (Wilson and Porras 1983, Butterfield et al 1997). All exotic reptiles and amphibians in Florida originally established in disturbed sites. However, several species have since spread to natural areas — these include *E. planirostris* (greenhouse frog), *Osteopilus septentrionalis* (Cuban treefrog) and *A. sagrei* (brown anole) (Butterfield et al 1997) and *Python molurus* (Fred Kraus personal communication).

Increasing levels of habitat disturbance may be creating more suitable habitat conditions for the establishment and spread of exotic reptiles and amphibians (Shine et al 2000). For example, the favoured habitat of the toad *R. catesbeiana*, which is native to eastern United States but is introduced in Colorado, is permanent lowland lakes and ponds. These habitats are not natural to Colorado but are becoming widespread through human activities, and this is creating suitable habitat for *R. catesbeiana* (Hammerson 1982). Similarly, African clawed frogs (*X. laevis*) have been introduced to Chile, and are found at higher densities in artificial water bodies (ponds, dams and irrigation channels) than in natural ponds or streams, although they are sometimes found in natural watercourses. They spread through agricultural areas using irrigation canals and Chile's expanding irrigated viticulture industry could aid the spread of *X. laevis* (Lobos and Jaksic 2005). According to Tinsley and McCoid (1996), being commensal with people has also helped *X. laevis* to expand its range in disturbed areas in California. In Florida, the expansion of the Miami metropolitan area is simultaneously destroying the preferred habitats of the native southern toad *Bufo terrestris* but creating new habitat for the cane toad *B. marinus.* Wilson and Porras (1983) found that the Cuban treefrog *O. septentrionalis* rapidly increased its range in urban areas in southern Florida, and suggest this spread was facilitated by urban swimming pools.

Petren and Case (1998) found human structural alterations to the environment facilitate invasion by geckos, by reducing interspecific competition between *Hemidactylus frenatus* and *Lepidodactylus lugubris*. Cole et al (2005) suggests that being anthropophilic contributes to the ability of *H. frenatus* to colonise locations outside its natural range.

Campbell and Echternacht (2003) consider habitat disturbance and fragmentation promote invasion success and suggest that an adaptation to open and disturbed habitats is one of the characteristics that contributes to the successful invasion history of the brown anole *A. sagrei*. Gorman et al (1978) found in general that exotic *Anolis* lizards behave like weeds that are commensal with people. For example, they found native populations of *A. richardi* on Grenada are widespread in both natural and disturbed conditions throughout a variety of habitats, and encompassing essentially the full altitudinal range of the island. In contrast, Gorman et al (1978) never found *A. richardi* in natural forested situations on Tobago, where it is introduced. On Tobago, *A. richardi* tend to abound in coconut groves and backyards.

The presence of other co-evolved exotic plants or animals may enhance the chances of establishment, by providing suitable food or shelter for an exotic species, or protection from predators. For example, Adams et al (2003) found that invasion of bullfrogs (*R. catesbeiana*) is facilitated by the presence of co-evolved non-native fish, which increase tadpole survival by reducing predatory macro-invertebrate densities. Native dragonfly nymphs in Oregon in the United States caused zero survival of bullfrog tadpoles in a replicated field experiment, unless a non-native sunfish (*Lepomis macrochirus*) was present to reduce dragonfly density. This pattern was also evident in pond surveys where the best predictors of bullfrog abundance were the presence of non-native fish and bathymetry (water depth relative to sea level). Kraus and Cravalho (2001) suggest that the dense populations of exotic prey species in Hawaii would make it easy for exotic snakes to establish there.

Mautz (cited in Tummons 2003) suggests the invasion of exotic frogs in Hawaii may have been facilitated by previous invasions. He suggests introduced nitrogen-fixing trees, particularly albizia, that are much more productive than the native `ohi`a dominated forests, provided a high-energy food source. This allowed an increased abundance of insects and other arthropods, which in turn 'set the stage' for invasion by coqui and other exotic frog species.

Risk assessment significance: Because many ecologists consider a species' ability to live in disturbed habitats increases the probability of its establishment, and because most successfully established exotic vertebrates are human commensals, this variable could be considered as a possible contributory factor when assessing the risk that new exotic species could establish in Australia. However, it is necessary to recognise that while environmental disturbance may enhance probability of success, it is possible for exotic reptiles and amphibians that can live in disturbed environments to also establish in undisturbed areas.

(iii) Suitable site — presence of resources and absence of enemies

The availability of habitat near the release site that meets a species' physiological and ecological needs is important for establishment. An absence or low occurrence of natural enemies such as predators, parasites, diseases or competitors is often suggested to favour establishment (Bomford 2003).

Case and Bolger (1991a, b) examined introduction success rates for exotic reptiles (primarily lizards) on Pacific islands and found communities with a rich reptile fauna were more resistant to invasion by exotic reptiles than communities with fewer reptile species. These authors present evidence supporting the hypothesis that predation and competition set important constraints on the distribution, colonisation (establishment) and abundance of reptiles (predominantly lizards) on islands. This evidence was based on studies of introduced exotics on Pacific islands and manipulative experiments.

In contrast, Rodda et al (2001) found introduced *Hemidactylus* gecko species are present on both Guana Island in the Carribean and on Guam in the Pacific. They also found that the failure of introduced *H. mabouia* to proliferate away from human habitation on Guana Island was unrelated to the presence of native lizard competitors (nocturnal predators), since none are known from the island. This example suggests caution in invoking competition to explain the abundance or distribution of *H. frenatus* in the Pacific as suggested by Case and Bolger (1991a, b) and Case et al (1994).

Losos et al (1993) reviewed data on 23 non-native *Anolis* introductions and concluded that the presence or absence of an ecologically similar native species was significantly correlated with colonisation success or failure. The presence of an ecologically similar species, a potential competitor, was often a factor in the failure of an introduced anole to establish. Powell et al (1990) found in the West Indies that where introduced *A. porcatus* occurred, its ecological analogue, the native *A. chlorocyanus* was uncommon or absent and conversely, where *A. chlorocyanus* was common, *A. porcatus* was apparently absent. Introduction in a locality of *A. porcatus* led to a decline in *A. chlorocyanus*. The introduced species appears to be more common in significantly disturbed urban habitats, whereas the native remains common in more complex habitats. These observations suggest competition occurs between the two species and that habitat disturbance facilitated the establishment and spread of the exotic species.

Meshaka (1997) suggests that the presence of an introduced predator anole (*A. equestris*) could hinder the establishment of exotic *A. porcatus* in southern Florida.

Rodda et al (1999) suggest the abundance of snake food on Guam probably accounts for the successful establishment and spread of the exotic brown treesnake *Boiga irregularis* on this island. Their modelling suggests that prey abundance both on Guam and in the native range of *B. irregularis* is the most important ecological variable limiting the density of this species. Guam has a high abundance of food for small and medium-sized *B. irregularis*. Based on this estimation of environmental suitability, the authors predict that *B. irregularis* could also do well on other currently snake-free islands in the Marianas if they should become established there. The authors suggest the high suitability of Guam habitats for *B. irregularis* is attributable to the success on Guam of introduced prey species, especially the house gecko *H. frenatus* and the terrestrial skink *Carlia ailanpilai*. Other important prey items for the snake are introduced birds — especially chickens (*Gallus gallus*), francolins (*Francolinus francolinus*), drongos (*Dicrurus macrocercus*), tree sparrows (*Passer montanus*), rock doves (*Columba livia*) and turtle doves (*Streptopelia bitorquata)* — and rats (*Rattus tanezumi* and *Rattus norvegicus*) and native lizards (*Emoia caeruleocauda*, *Lepidodactylus lugubris* and *Gehyra mutilata*). The introduction and high populations of rats on Guam before the arrival of *B. irregularis,* and the irruptions of shrews (*Suncus murinus*) and skinks (*Carlia ailanpilai*) accelerated the population expansion of the brown treesnake. Were it not for the highly successful establishment of introduced prey species, Guam would probably not now have such a dense population of *B. irregularis*.

Risk assessment significance: No consistent patterns between community structure and susceptibility to invasion have been demonstrated for exotic reptiles and amphibians. Therefore, variables describing the biotic components of recipient habitats are unlikely to have predictive value, until such time as long-term intensive studies on community interactions in relation to the physiological and life history requirements of the species proposed for introduction are first conducted. The potential relationships between an organism and possible parasites, predators, diseases and competitors are usually impossible to predict, except in a generalised and qualitative sense. These factors are difficult or expensive to measure quantitatively, so there is little evidence to support or reject their role in establishment success. Hence, these factors are unlikely to be of value for risk assessment and management. It would also be extremely difficult to objectively rank these biotic components of habitat suitability. Hence, this factor probably has limited value for quantitative risk assessment except for separating disturbed habitat from undisturbed habitat. The significance of the availability of suitable microhabitats and microclimates for exotic reptiles and amphibians is largely unknown.

(iv) Broad diet

Species with a broad diet (dietary generalists) may be more successful at establishing exotic populations than those with a restricted diet (dietary specialists) (Bomford 2003).

Cole et al (2005) suggest that being a generalist contributes to the ability of the gecko *H. frenatus* to colonise locations outside its natural range. Da Silva and Blasco (1995) consider it likely that the broad ecological tolerances and omnivorous diet of red-eared sliders (*T. scripta)* will contribute to this species establishing breeding populations in southwestern Spain. Wilson and Porras (1983) suggest that one reason for the success of *Anolis equestris* anole in urban areas of south Florida may be its broad diet — it eats palm, mango and *Ficus* fruit, azalea flowers, tree sap, leaves, caterpillars, large ants, spiders, leafhoppers, cicadas, cockroaches, beetles, tree frogs, smaller anoles, young birds, young rodents. Campbell and Echternacht (2003) suggest that the generalised diet of the brown anole *A. sagrei* is one of the characteristics that contribute to its successful invasion history. None of these authors present any evidence to support their speculations.

Risk assessment significance: Because many ecologists consider that having a generalist diet increases the probability of establishment success, and because nearly all exotic vertebrates established in Australia do have generalist diets, this variable might be considered as a possible contributory factor for assessing the risk that new exotic species could establish here. However, given nearly all reptiles and amphibians do have generalist diets, this factor is unlikely to be of much practical use for discriminating between species which have a high or low risk of establishing in Australia.

(v) Generalists — behaviour, habitat use, adaptability

Behavioural generalists and species with high adaptability may be more successful than specialists (Bomford 2003).

Wilson and Porras (1983) suggest that one reason for the success of *A. sagrei* in southeast Florida is its broad adaptability in edificarian areas; that is, in habitats dominated by buildings, with little vegetation. Wilson and Porras (1983) also suggest that one reason for the success of the spiny-tailed iguana (*Ctenosaura pectinata*) in Florida is the range of habitats it lives in including piles of building boards, piles of tree trunks and branches, rock walls, roofs and foundations of houses, trash piles and tree hollows.

Campbell and Echternacht (2003) suggest that the geographic variability of the native range of *A. sagrei* and its generalised habitat use are two of the characteristics that contribute to the successful invasion history of this anole. For example, in the Bahamas *A. sagrei* exhibits rapid morphological changes in response to local conditions and in Florida it exhibits high levels of geographic variability in some morphological characteristics.

Risk assessment significance: Although many ecologists consider being an adaptable generalist with broad habitat preferences may contribute to the invasiveness of exotic species, this factor has been little studied for exotic reptiles and amphibians. Measuring and quantifying a species' 'adaptability' and 'generalism' would be difficult. Therefore, this factor is probably only useful in a broad qualitative sense for assessing the risk that exotic reptiles and amphibians could establish exotic populations in Australia.

(vi) Rate of population increase and related variables

Some ecologists consider that high fecundity (average number of female offspring produced that survive to reproductive age) and associated attributes (early sexual maturity, large clutch size, high breeding frequency, short gestation and opportunistic breeding) contribute to successful vertebrate invasions (Bomford 2003).

Wilson and Porras (1983) suggest that one reason for the success of *A. equestris* in urban areas of south Florida may be its longevity relative to most other anoline species. According to McCoid and Fritts (1993, 1995) *X. laevis* frogs in California have an extended breeding season, year-long growth and maturation in as little as eight months and these authors consider these factors are the prime reason for the frog's rapid establishment and continued range expansion in California.

Risk assessment significance: The evidence supporting a link between factors associated with a high fecundity or rates of population increase and high establishment success is limited and equivocal for vertebrates generally and none could be found for exotic reptiles and amphibians. Therefore, it is unlikely that factors associated with rate of increase will be useful at present for predicting the probability of establishment success.

(vii) Single female able to colonise alone

Kraus and Cravalho (2001) suggest the likelihood of establishment may be increased by the ability of some common snake species (such as boas) that normally reproduce sexually, to facultatively reproduce parthenogenically in the prolonged absence of males.

A number of lizard species, such as *Lepidodactylus lugubris*, and one snake, *Ramphotyphlops braminus*, consist entirely of females and are obligately parthenogenic. This makes it theoretically possible for a single, unimpregnated female to establish an exotic population.

Campbell and Echternacht (2003) suggest that an ability to store sperm is one of the characteristics that contributes to the successful invasion history of the brown anole (*A. sagrei*).

Risk assessment significance: There is no evidence that species that can colonise from a single individual have higher introduction success. However, it is possible that such species have a lower minimum viable propagule size than others, so there may be a higher risk of such species establishing.

(viii) Dispersal ability

Da Silva and Blasco (1995) suggest it is likely that the dispersal ability of *T. scripta* turtles will contribute to this species establishing breeding populations in southwestern Spain. Campbell and Echternacht (2003) suggest that an ability to disperse directly across water is one of the characteristics that contributes to the successful invasion history of the brown anole (*A. sagrei*).

Risk assessment significance: Dispersal ability is generally a difficult trait to quantify. It is likely that good dispersal ability has increased the frequency of introduction of some species. Dispersal ability is also likely to affect rate of spread following establishment. However, dispersal ability has been little examined as a risk factor for establishment success, so it is currently unlikely to be useful for predicting the probability of establishment.

(ix) Island introductions more successful than mainland introductions

Butterfield et al (1997) suggested islands are more vulnerable to exotic invasions by reptiles and amphibians than mainlands. Kraus (2003) examined published introduction records of exotic reptiles and amphibians around the world. He found that more introductions (n = 316) occur on islands than on continents (n = 226), and 72% of island introductions led to successful establishment compared to 60% on continents. While these data show introductions to islands are more successful, this is probably due at least in part to the introduction pathway. Most introductions to islands occurred via cargo shipping, whereas those to continents primarily involved the pet trade. Over all world introductions, those made via cargo have a 54% success rate whereas those involved with the pet trade have a 47% success rate.

Risk assessment significance: Islands may be slightly more vulnerable than continents to invasion by exotic reptiles and amphibians. Further analyses of world introduction records would be required to determine whether this factor can be used to better inform risk assessments for establishment success.

(x) Body mass

Animals with higher body mass may be more successful at establishing exotic populations than lighter, related species (Ehrlich 1986, 1989). On the other hand, Cole et al (2005) suggest that small body size contributes to the ability of the gecko *H. frenatus* to colonise locations outside its natural range.

Risk assessment significance: Body mass has been little examined as a risk factor for establishment success, so it is currently unlikely to be useful for predicting the probability of establishment for reptiles and amphibians.

3.3 Risk assessment for establishment of exotic reptiles and amphibians introduced to Australia

Exotic reptiles and amphibians have a world introduction success rate of 51.5% (Bomford et al 2008, Kraus in press). Too few reptiles and amphibians have been introduced to Australia (Kraus in press, unpublished database) to enable quantitative comparisons between successful and failed species. Only five successful species and two failed species are known for mainland Australia (Appendix C, Table C2). Therefore, Bomford et al (2005) and Bomford (2006) assumed that the factors affecting establishment success for reptiles and amphibians introduced to Britain and the United States would also apply to reptiles and amphibians introduced to Australia.

Establishment Risk Scores for reptiles and amphibians introduced to Australia are the sum of three risk scores:

1. Climate Match Score
2. Exotic Elsewhere Score
3. Taxonomic Family Score.

Instructions for calculating these scores are presented in Section 3.4. A species' Establishment Risk Score is converted to an Establishment Risk Rank ranging from Low to Extreme as described (Section 3.4). Note the Establishment Risk Score cut-off threshold between the Low and Moderate Establishment Risk Rankings has been adjusted upwards from Bomford's (2006) value to bring the proportion of successful species in the Low and Moderate risk ranks into line with the other seven risk assessment models presented in this report. The cut-off thresholds for the other risk ranks remain unchanged.

The numbers of species in each Establishment Risk Rank are presented in Figure 3.2. This figure shows that the Establishment Risk Ranks for exotic reptiles and amphibians introduced to Britain, Florida and California strongly predict introduction outcomes.

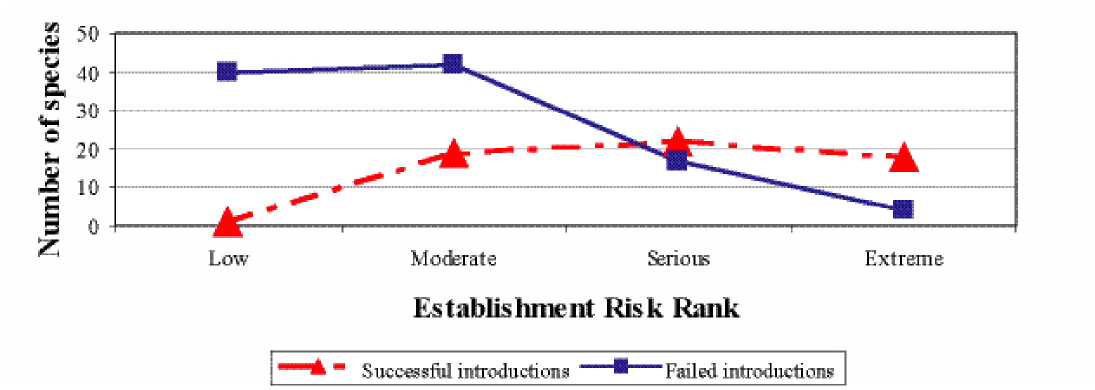

Figure 3.2 Number of species in each Establishment Risk Rank for reptiles and amphibians (combined) introduced to Britain, California and Florida (combined)
Establishment Risk Scores and Ranks were calculated using the directions given in Section 3.4.

The data from Britain and the United States were used by Bomford et al (2005) and Bomford (2006) to develop a model to rank the relative establishment risks of species proposed for import to Australia. But the low number of species introduced to Australia meant there was no quantitative basis for determining cut-off thresholds for discriminating between high, moderate and low-risk species. To overcome this problem, Bomford et al (2005) and Bomford (2006) assumed that risk threshold values for Establishment Risk Scores for the combined Britain, California and Florida dataset would translate to equivalent levels of establishment risk for Australia. This is an untested assumption.

As Bomford et al (2008) found that jurisdiction significantly affected establishment success for exotic reptiles and amphibians, these cut-off thresholds based on averaged values may not be accurate for Australia. However, it is hoped that the large total sample size and variable conditions in the three jurisdictions used will give some robustness to the cut-off thresholds presented in Section 3.4 for the Australian reptile and amphibian model. This model does give reasonable predictions for the seven exotic reptile and amphibian species known to have been introduced to Australia (Appendix C, Table C2). The model gives one successful species (cane toad *B. marinus*) an Establishment Risk Rank of Extreme, and gives the other four successful species ranks of Serious. For the two failed species, the model ranks the Establishment Risk Rank of one (axolotl *Ambystoma mexicanum*) as Low but the other (black-spined toad *B. melanostictus*) is ranked as Serious. However, *B. melanostictus* has only been detected in very low numbers at Australian ports, and it is possible that it is a high-risk species that has not yet been subjected to sufficient propagule pressure, or that has been detected and killed before being able to establish.

The reliability of predictions made by the model presented in this section is uncertain for Australia because the cut-off thresholds used in Figure 3.2 are untested. Therefore, we provide an alternative model: Section 3.5 adapts Bomford's (2003) Bird and Mammal Model (Model 1 in Section 2.5) for use in assessing establishment risk for exotic reptiles and amphibians proposed for introduction to Australia. Exotic reptiles and amphibians can then be assessed using both models. If both models predict an equivalent level of risk, then that result may be more robust than the result taken from Bomford et al's (2005) model alone. If the two models predict different levels of risk, a precautionary approach would accept the higher level of risk.

Bomford et al (2008) developed a generalised linear mixed model to describe probability of establishment success for reptiles and amphibians introduced to Britain, California and Florida.

Because these authors found that jurisdiction had a significant effect, it is not possible to use their model to calculate a precise probability value for establishment for species introduced to other jurisdictions. However, their model can be used to more generally rank species' risk of establishment from high risk to low risk for other jurisdictions. Instructions for using an adapted version of Bomford et al's (2008) model for ranking establishment risk for species introduced to Australia are presented in Section 3.6.

<u>Reptile and Amphibian Model</u> (Bomford et al 2008):

Bomford et al's (2008) model for the probability of establishment of exotic reptiles and amphibians is:

P(Establishment) = 1/(1 + exp(3.8499 – 2.9016(*prop.species*) – *Jurisdiction score* – *S (Climate 6)* – *Family random effect*).

P(Establishment) = probability of establishment.

Prop.species = number of jurisdictions where species successfully established divided by the total number of jurisdictions where species introduced.

Jurisdiction score = 0 for Britain, 2.4702 for California and 3.0488 for Florida.

S(Climate 6) = a smooth function of the climate match score expressed as a proportion of all data locations in the jurisdiction (Note: instructions for calculating this variable are presented in Section 3.6).

Family random effect = a family random effect assumed drawn from a Gaussian distribution with mean zero and variance that was estimated from Bomford et al's (2008) data. (Note: a table listing of these values is presented in Section 3.6).

P(Establishment) values for exotic reptiles and amphibians introduced to Britain, Florida and California calculated using Bomford et al's (2008) model are presented in Appendix C, Table C3. *P(Establishment)* values are converted to an Establishment Risk Ranks ranging from Low to Extreme for each species (Figure 3.3). This figure shows that the Establishment Risk Ranks calculated from *P(Establishment)* values from Bomford et al's (2008) model for reptiles and amphibians introduced to Britain, Florida and California strongly predict introduction outcomes.

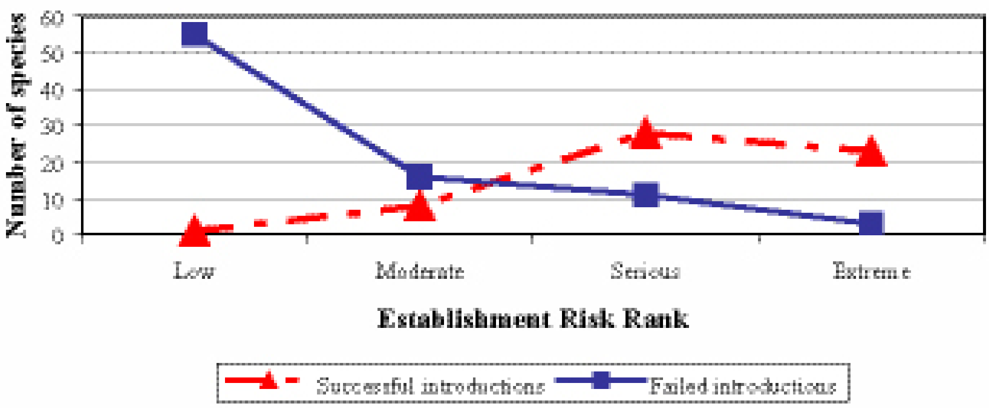

Figure 3.3 Number of species in each Establishment Risk Rank for reptiles and amphibians introduced to Britain, California and Florida (combined)
P(Establishment) values were calculated using the model developed by Bomford et al (2008) and then converted to four Establishment Risk Ranks using the directions given in Section 3.6.

Bomford et al's (2008) model did not include data for Australia and therefore no Jurisdiction score is available for Australia. In the risk assessment model presented in Section 3.6 (which adapts from Bomford et al's 2008 model for reptile and amphibian species proposed for import to Australia), Florida's Jurisdiction score is used. This value was selected because it gives the cane toad an Establishment Risk Rank of Extreme for Australia, when the cut-off thresholds presented in Section 3.6 are used. However, because this Jurisdiction score is not validated for Australia, the Establishment Risk Scores calculated for Australia in Section 3.6 are not estimates of probability of establishment. Rather, they provide a relative ranking of establishment risk for exotic reptile and amphibian species introduced to Australia. The model presented in Section 3.6 can only be used for species in families that were included in Bomford et al's (2008) model. Results for the species used in Bomford et al's (2008) model are presented in Appendix C, Table C4.

3.4 Instructions for using the Australian Reptile and Amphibian Model

The model presented in this section is the original model published by Bomford et al (2005), modified by Bomford (2006) to give a four-rank risk outcome instead of the original six-rank outcome. The model applies to the Australian mainland and Tasmania but not to small offshore islands.

A. Climate Match Risk Score

Use PC CLIMATE (Bureau of Rural Sciences 2006) or CLIMATCH (Bureau of Rural Sciences; see http://www.brs.gov.au/climatch) and select:

- *'worlddata_all.txt'* as the world data location
- all 16 climatic parameters for matching locations (see Table 1)
- *'Euclidian match'* for the analysis
- Splined (gridded) surface for Australian 'match to' file.

Map the selected reptile or amphibian species' overseas range, including its entire native and exotic (excluding Australia) ranges over the past 1000 years in CLIMATE, to use as the species' input range.

Score A = A species' Climate Match Risk Score = the sum of its four scores for Euclidian match classes 7–10 (that is Sum level 7) expressed as a percentage of the maximum possible score for all these classes (that is 2785 for Australia).

If the input area has 12 or fewer meteorological stations, then CLIMATE is likely to underestimate the climate match to Australia. If this is the case, it is advisable to increase the Climate Match Risk Score by ten percentage points.

Example 1: The cane toad (*Bufo marinus*) scores:

Sum Euclidian match scores to Australia levels 7–10 = 1848

Score A = Climate Match Risk Score = 100 × (1848/2785) = 66

Example 2: A lizard has only eight meteorological stations in its overseas range and the sum of its four highest Euclidian match classes Sum Level 7 = 362. Its Climate Match Risk Score (Score A) = 100 × (362/2785) + 10 = 13 + 10 = 23.

B. Exotic Elsewhere Risk Score

Score B = A species' Exotic Elsewhere Risk Score, calculated as follows:

- Species has established a breeding self-sustaining exotic population in another country = 30.
- Species has been introduced into another country and records exist of it in the wild, but it is uncertain if a breeding self-sustaining exotic population has established = 15.
- Species has not established an exotic population (including species not known to have been introduced anywhere) = 0.

For example, the cane toad gets a Score B of 30 for Australia because it has established self-sustaining exotic populations in many overseas countries including in Asia, Africa and many Pacific islands.

C. Taxonomic Family Risk Score

Score C = A species' Taxonomic Family Risk Score, taken from Table 3.1 below.

Table 3.1 Taxonomic Family Risk Scores for exotic reptiles and amphibians
(Based on data sourced from F. Kraus, unpublished database).

Family	Successful introduction events %	Taxonomic Family Risk Score
Agamidae	70	30
Alligatoridae	15	10
Ambystomatidae	38	15
Amphisbaenidae	0	0
Anguidae	29	10
Boidae	6	5
Bufonidae	60	20
Chamaeleonidae	79	30
Chelidae	22	10
Chelydridae	29	10
Colubridae	20	10
Cordylidae	17	10
Crocodylidae	0	0
Cryptobranchidae	0	0
Dendrobatidae	100	30
Discoglossidae	38	15
Elapidae	11	10
Emydidae	39	15
Gekkonidae	76	30
Geomydidae	0	0
Gymnophthalmidae	0	0
Helodermatidae	0	0
Hylidae	41	15
Iguanidae	56	20
Kinosternidae	0	0

Family	Successful introduction events %	Taxonomic Family Risk Score
Lacertidae	57	20
Leptodactylidae	79	30
Microhylidae	60	20
Myobatrachidae	40	15
Pelobatidae	0	0
Pelomedusidae	25	10
Pipidae	42	15
Plethodontidae	58	20
Proteidae	100	30
Pygopodidae	0	0
Ranidae	80	30
Rhacophoridae	75	30
Salamandridae	36	15
Scincidae	46	15
Teiidae	67	20
Testudinidae	48	15
Trionychidae	66	20
Typhlopidae	95	30
Varanidae	38	15
Viperidae	21	10

For example, the cane toad is in Family Bufonidae so has a Taxonomic Family Risk Score of 20.

Establishment Risk Score

A species' Establishment Risk Score = Score A + Score B + Score C.

For example, the cane toad's Establishment Risk Score for Australia =
66 + 30 + 20 = 116.

Establishment Risk Rank

A species' Establishment Risk Score can be converted to an Establishment Risk Rank ranging from Low to Extreme, using the following cut-off thresholds:

Establishment Risk Rank	Establishment Risk Score
Low	≤22
Moderate	23–60
Serious	61–115
Extreme	≥ 116

For example, the cane toad's Establishment Risk Score for Australia is 116 = Extreme.

3.5 Instructions for using the Bird and Mammal Model for reptiles and amphibians

The model applies to the Australian mainland and Tasmania but not to smaller offshore islands.

A. Climate Match Score (1–6)

Map the selected reptile or amphibian species' overseas range, including its entire native and exotic (excluding Australia) ranges over the past 1000 years. Use PC CLIMATE (Bureau of Rural Sciences 2006) or CLIMATCH (Bureau of Rural Sciences; see http://www.brs.gov.au/climatch), to determine the climate match between this overseas range and Australia, selecting Closest Standard Match and using all 16 climate variables for the analysis.

Sum the values for the five highest match classes (ie the scores for match classes 10, 9, 8, 7 and 6) Sum level 6 = 'Value X'.

Convert Value X to a Climate Match Score using the following cut-off thresholds:

CLIMATE Closest Standard Match Sum Level 6 (Value X)	Climate Match Score
<100	1
100–599	2
600–899	3
900–1699	4
1700–2699	5
≥ 2700	6

If the input range for a species has 12 or fewer meteorological stations, then it is likely to underestimate the climate match to Australia. If this is the case, it is advisable to increase the climate match score by one increment. For example, if the input range for a species included only five meteorological stations, and the sum of the values for the five highest match classes to Australia equalled 504 (ie Value X = 504), then this would give a Climate Match Score of 2 + 1 = 3.

B. Exotic Population Established Overseas Score (0–4)

This score is calculated as follows:

- No exotic population ever established = 0.
- Exotic populations only established on small island (< 50 000 km^2; Tasmania is 67 800 km^2) = 2.
- Exotic population established on a larger island (> 50 000 km^2) or anywhere on a continent = 4.

C. Overseas Range Size Score (0–2)

Calculate Overseas Range Size Score based on an estimate of the species' overseas range size (including current and past 1000 years, natural and introduced range) in millions of square kilometres using the following cut-off thresholds:

Overseas Range Size Score	Overseas range size (million km^2)
0	0–1
1	2–69
2	≥ 70

Establishment Risk Score (1–12)

Calculate the Establishment Risk Score as the sum of the above three scores = A + B + C.

Establishment Risk Rank

Convert the Establishment Risk Score obtained above to an Establishment Risk Rank (Low, Moderate, Serious or Extreme) using the following cut-off thresholds:

Establishment Risk Rank	Establishment Risk Score
Low	≤ 4
Moderate	5–7
Serious	8–9
Extreme	10–12

3.6 Instructions for using Bomford et al's (2008) Reptile and Amphibian Model to rank establishment risk for exotic reptiles and amphibians introduced to Australia

The model used to rank risk of establishment presented in this section is based on the analyses by Bomford et al (2008) for exotic reptiles and amphibians introduced to Britain, California and Florida. Hence, some parameter values required for using the model are only available for the taxa that had been introduced to these jurisdictions. The model applies to the Australian mainland and Tasmania, but not to small offshore islands.

A. Family Random Effect value

Family Random Effect values are only available for the families of species that were used in Bomford et al's (2008) analysis of species introduced to Britain, California or Florida. These values are presented in Table 3.2.

Table 3.2 Family random effect values for reptiles and amphibians introduced to Britain, California or Florida

Family	*Family Random Effect*	Family	*Family Random Effect*
Agamidae	-0.11	Bufonidae	-0.91
Alligatoridae	0.43	Chamaeleonidae	0.48
Alytidae	0.01	Chelydridae	0.68
Ambystomatidae	0.64	Colubridae	-0.15
Boidae	-0.09	Elapidae	0.74
Bombinatoridae	-0.26	Emydidae	-0.77

Family	*Family Random Effect*
Gekkonidae	-0.41
Geoemydidae	-0.37
Hylidae	-0.82
Iguanidae	0.49
Kinosternidae	-0.05
Lacertidae	0.34
Leptodactylidae	1.08
Pelobatidae	-0.22
Pelomedusidae	-0.62
Pipidae	0.03

Family	*Family Random Effect*
Pythonidae	-0.08
Ranidae	1.69
Salamandridae	0.35
Scincidae	0.00
Teiidae	-0.77
Testudinidae	-1.30
Trionychidae	-0.07
Typhlopidae	0.02
Varanidae	-0.59

B. Prop.species value

The *Prop.species* value is the number of jurisdictions where species successfully established divided by the total number of jurisdictions where species has been introduced. A jurisdiction is either a country or for North America, is a state or province, or is a major island or island group that is part of a larger country (eg Galapagos, Ryukyus). *Prop.species* values can only be calculated reliably for species that have already been introduced to at least three jurisdictions outside their native range. Table 3.3 presents *Prop.species* values for reptiles and amphibians calculated from data taken from a global database of alien herpetological introductions, which will be published (updated) in its entirety (Kraus in press). Once Kraus' updated data are published, they should be used in preference to the values presented in Table 3.3.

Table 3.3 Prop.species values for reptiles and amphibians for which there are world records for introductions to three or more jurisdictions worldwide
Values were calculated from data taken from a global database of alien herpetological introductions that will be published in a slightly updated form in its entirety (Kraus in press).

Species	*Prop.species value*
Agama agama	0.50
Agkistrodon piscivorus	0.00
Aldabrachelys gigantea	0.60
Alligator mississippiensis	0.00
Alytes obstetricans	0.50
Ambystoma mexicanum	0.00
Ambystoma tigrinum	0.44
Anguis fragilis	0.33
Anolis carolinensis	0.67
Anolis cristatellus	1.00
Anolis distichus	0.67
Anolis equestris	0.67

Species	*Prop.species value*
Anolis extremus	0.80
Anolis sagrei	0.61
Apalone spinifera	0.57
Boa constrictor	0.15
Boiga irregularis	0.22
Bombina bombina	0.20
Bombina orientalis	0.00
Bufo bufo	0.29
Bufo gutturalis	1.00
Bufo marinus	0.69
Bufo melanostictus	0.50
Bufo viridis	0.60

Species	Prop.species value
Caiman crocodilus	0.25
Calotes versicolor	0.89
Carlia ailanpalai	1.00
Chalcides ocellatus	0.33
Chamaeleo chamaeleon	0.75
Chelydra serpentina	0.31
Chondrodactylus bibronii	0.67
Chrysemys picta	0.09
Cnemidophorus lemniscatus	1.00
Crotalus viridis	0.20
Cuora flavomarginata	0.33
Cynops pyrrhogaster	0.00
Cyrtopodion scabrum	1.00
Diadophis punctatus	0.00
Discoglossus pictus	0.67
Drymarchon corais	0.00
Elaphe guttata	0.12
Elaphe obsoleta	0.17
Eleutherodactylus coqui	0.70
Eleutherodactylus johnstonei	0.89
Eleutherodactylus planirostris	0.91
Emydoidea blandingi	0.00
Emys orbicularis	0.44
Epicrates cenchria	0.00
Gehyra mutilata	0.72
Gekko gecko	0.33
Glyptemys insculpta	0.00
Gonatodes albogularis	0.86
Gopherus agassizii	0.00
Gopherus berlandieri	0.00
Gopherus polyphemus	0.00
Graptemys geographica	0.33
Graptemys pseudogeographica	0.10
Hemidactylus brookii	0.82
Hemidactylus flaviviridis	1.00
Hemidactylus frenatus	0.88
Hemidactylus garnotii	0.70
Hemidactylus mabouia	1.00

Species	Prop.species value
Hemidactylus turcicus	0.87
Hemiphyllodactylus typus	0.75
Hierophis viridiflavus	0.67
Hyla arborea	0.00
Hyla cinerea	0.50
Hyla meridionalis	0.50
Iguana iguana	0.65
Kaloula pulchra	0.67
Kinosternon subrubrum	0.00
Lacerta bilineata	0.50
Lampropeltis getula	0.00
Lampropeltis triangulum	0.00
Lampropholis delicata	1.00
Laudakia stellio	0.67
Lepidodactylus lugubris	0.80
Leptodactylus fallax	0.00
Leptodeira annulata	0.00
Lipinia noctua	0.88
Lissemys punctata	0.20
Litoria aurea	0.83
Litoria caerulea	0.00
Litoria ewingii	0.50
Litoria fallax	0.33
Lycodon aulicus	0.75
Lygosoma bowringii	0.67
Mabuya aurata	1.00
Mabuya multifasciata	0.67
Macroclemys temminckii	0.00
Malaclemmys terrapin	0.00
Mauremys caspica	0.00
Mauremys leprosa	0.17
Mauremys mutica	0.50
Mauremys reevesii	0.17
Mediodactylus kotschyi	0.67
Micrurus fulvius	0.00
Natrix maura	0.33
Natrix natrix	0.00
Natrix tessellata	0.17

Species	Prop.species value
Necturus maculosus	1.00
Ocadia sinensis	0.00
Ophisaurus apodus	0.00
Osteopilus septentrionalis	0.65
Palea steindachneri	0.50
Pelodiscus sinensis	0.67
Pelusios subniger	0.40
Phelsuma cepediana	1.00
Phelsuma dubia	1.00
Phelsuma laticauda	1.00
Phrynosoma cornutum	0.21
Pleurodeles waltl	0.00
Podarcis dugesii	0.50
Podarcis muralis	0.82
Podarcis pityusensis	0.60
Podarcis sicula	0.75
Polypedates leucomystax	0.67
Proteus anguinus	0.67
Pseudacris regilla	0.50
Pseudemys concinna	0.25
Pseudemys floridana	0.00
Pseudemys nelsoni	0.25
Ptychadena mascareniensis	0.60
Python molurus	0.13
Python regius	0.00
Python reticulatus	0.00
Python sebae	0.00
Ramphotyphlops braminus	0.97
Rana berlandieri	1.00
Rana catesbeiana	0.87
Rana clamitans	0.50
Rana esculenta	0.50
Rana grylio	0.67
Rana nigromaculata	0.60
Rana perezi	1.00

Species	Prop.species value
Rana pipiens	0.46
Rana ridibunda	0.90
Rana rugosa	1.00
Rana sphenocephala	0.50
Rana sylvatica	0.33
Rana temporaria	0.50
Salamandra salamandra	0.00
Scinax rubra	0.60
Speleomantes ambrosii	1.00
Sphaerodactylus argus	1.00
Tarentola mauritanica	0.71
Terrapene carolina	0.06
Terrapene ornata	0.00
Testudo graeca	0.38
Testudo hermanni	0.36
Testudo marginata	0.40
Thamnophis sirtalis	0.00
Timon lepidus	0.00
Trachemys scripta	0.66
Trachemys stejnegeri	0.57
Triturus alpestris	0.75
Triturus carnifex	0.75
Triturus vulgaris	0.33
Tupinambis teguixin	0.33
Uromastyx acanthinurus	0.00
Varanus exanthematicus	0.00
Varanus indicus	1.00
Varanus niloticus	0.33
Varanus salvator	0.00
Vipera ammodytes	0.33
Xenopus laevis	0.47
Zootoca vivipara	0.00

C. *S(Climate 6)* value

Use CLIMATE (Bureau of Rural Sciences 2006) or CLIMATCH (Bureau of Rural Sciences; see http://www.brs.gov.au/climatch) and select:

- *'worlddata_all.txt'* as the world data location
- all 16 climatic parameters for matching locations (see Table 1.1)
- *'Euclidian match'* for the analysis
- Splined (gridded) surface for Australian 'match to' file.

Map the selected reptile or amphibian species' overseas range, including its entire native and exotic (excluding Australia) ranges over the past 1000 years in CLIMATE, to use as the species' input range.

Perform a Euclidian match and then calculate the sum of the five scores for classes 6–10. Express this as a proportion of the maximum possible score (that is 2785 for Australia).

Look up the Climate 6 score along the x-axis of Figure 3.4. Read off the y-axis the equivalent *S(Climate 6)* value.

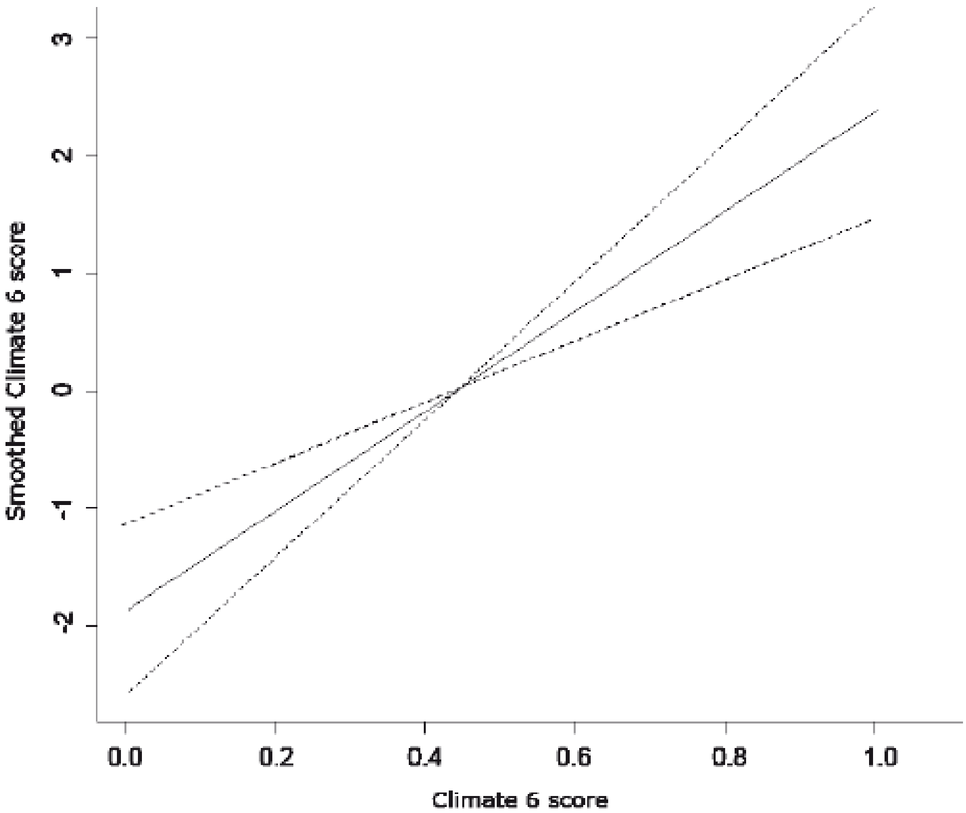

Figure 3.4 Penalised regression spline fit of Climate 6 score for reptiles and amphibians
The solid line (fitted by the model) indicates that as the Climate 6 score increases (x-axis) the chance of successful introduction increases. The dotted lines indicate the 95% confidence interval around the line. The solid line is used to convert raw Climate 6 scores to smoothed *S(Climate 6)* scores. The equation for this line is *S(Climate 6)* = 4.25(Climate 6 Score) – 1.88.

For example, the cane toad (*B. marinus*) scores as follows:

Sum Euclidian match scores to Australia levels 6–10 = 2370

Climate 6 score for the cane toad = 2370/2785 = 0.85

S(Climate 6) value for the cane toad from Figure 3.4 = 1.73.

Establishment Risk Score

Establishment Risk Score = 1/(1 + exp(0.80 – 2.90(*Prop.species*) – *S(Climate 6)* – *Family Random Effect))*

Example 1. The cane toad *B. marinus* is in the family Bufonidae. So, the Establishment Risk Score for *B. marinus* = 1/(1 + exp(0.80 – 2.90(0.69) – 1.73 – (-0.91))

= 1/(1 + exp (0.80 – 2.90(0.69) – 1.73 + 0.91))

= 1/(1 + exp(-2.003))

= 1/(1+0.135)

= 0.88.

Example 2. The axolotl (or salamander) *Ambystoma mexicanum* is in the family Ambystomatidae and therefore its Family Random Effect value from Table 3.2 is 0.64. Its *Prop.species* value from Table 3.3 is 0.00. Its raw Climate Match Score Sum level 6 for Australia is 0. Therefore its Climate 6 score = 0/2785 = 0.00 and its *S(Climate 6)* value from Figure 3.4 = -1.85. Establishment Risk Score for *A. mexicanum* = 1/(1 + exp(0.80 – 2.90(0.00) + 1.85 – 0.64))

= 1/(1 + exp(2.01))

= 1/(1 + 7.46)

= 0.12.

Establishment Risk Rank

Establishment Risk Scores are converted to Establishment Risk Ranks using the following conversions:

Establishment Risk Rank	Establishment Risk Score
Low	≤0.16
Moderate	0.17–0.39
Serious	0.40–0.85
Extreme	≥ 0.86

Example 1. The cane toad has an Establishment Risk Score of 0.88, which gives it an Establishment Risk Rank of Extreme.

Example 2. The axolotl has an Establishment Risk Score of 0.12, which gives it an Establishment Risk Rank of Low.

Other species

Table 3.2 does not include Family Random Effect values for species that Bomford et al (2008) did not include in their analyses of introductions to Britain, California and Florida. For such species, a range of potential *P(Establishment)* values could be calculated by inserting the minimum (-1.3) and maximum (1.69) Family Random Effect scores from Table 3.2 into the model. These minimum and maximum *P(Establishment)* values could then be used to calculate the minimum and maximum Establishment Risk Rank(s) for the species.

*Prop.specie*s values in Table 3.3 were calculated from introduction records in Kraus's (in press) database. Only species for which there were three of more introduction records are included in this table. For species not included in these tables, data on successful and failed introduction records would need to be obtained from other sources. When Kraus' (in press) database is published, it will contain additional records for some species. However, Kraus' database only includes introduction records obtained from the published literature, and it may be possible to obtain additional reliable unpublished introduction records for some species. Also, excluded from Bomford et al's (2008) analyses and from Table 3.3 were introduction records for which the outcome (succeeded or failed to establish) is unknown or uncertain. A check of more recent data sources could clarify the outcome of some of these introduction events.

3.7 Factors affecting assessments of the pest status of introduced reptiles and amphibians

A number of factors affect assessments of the pest status of exotic reptiles and amphibians. These factors include the reliability of evidence of causing harm, confounding factors and the species' adaptation to its introduced environment. Examples from the literature are reviewed in this section.

3.7.1 Evidence reliability and impacts caused by confounding factors

Knowledge about the impacts of exotic reptiles and amphibians is poor and often anecdotal (Wilson and Porras 1983, Freeland 1984, Butterfield et al 1997, Lever 2003, Spinks et al 2003, Smith 2005a, b). Many of the impacts attributed to exotic reptiles and amphibians are correlative or anecdotal. Nonetheless, the diet and behaviour of some reptiles and amphibians definitely gives them the potential to harm native species and cause other environmental damage in their introduced habitats. This potential, combined with measured changes in abundance or distribution of vulnerable native species following the introduction of exotics to new habitats, provides compelling evidence of harmful impacts.

Reliable knowledge about impacts for most exotic reptiles and amphibians, both in Australia and overseas, is sparse for two main reasons. Firstly, there has been limited research, and preinvasion datasets in particular are usually scarce. Secondly, introductions of exotic reptiles and amphibians have often coincided with other changes. This means impacts due to exotic reptiles and amphibians are confounded with impacts due to other factors, making it difficult to determine the impacts of the exotic species. Some of these confounding factors, and examples found in the literature are listed below:

(i) Disturbance by people

Disturbance by people includes through habitat disturbance and destruction, urbanisation, pollution, altered water regimes, increasing pesticide residues, introductions of exotic plants and grazing by domestic stock.

The introduced red-eared slider *T. scripta* is thought to threaten the native pond turtle *Emys orbicularis* in Europe, but according to Luiselli et al (1997) other threats may play a role, including habitat loss, pollution and highway mortality.

Although introduced bullfrogs *R. catesbeiana* have been blamed for amphibian declines in much of western North America; additional causes may include water pollution and habitat disturbance (Hammerson 1982).

According to Wilson and Porras (1983), the introduced cane toad *B. marinus* is replacing the native southern toad *B. terrestris* in Florida and this has sometimes been attributed to competition between the two species. But Wilson and Porras (1983) suggest that *B. terrestris* has declined due to failure to adapt to human-caused changes to vegetation and water supply, and that this occurred before the invasion of *B. marinus*.

In Papua New Guinea, the Papuan black snake *Pseudechis papuanus* apparently declined around Port Moresby following the introduction of the cane toad *B. marinus.* This was possibly due to cane toad poisoning following attempts by the snake to eat toads. But the snake's decline may also have been due to other factors such as increasing urbanisation and traffic (Lever 2003).

The introduced African clawed frog *X. laevis* were found to have native tidewater gobies *Eucyclogobius newberryi* in their stomachs in brackish streams and estuaries in California. Tidewater gobies have declined and predation by *X. laevis* may have played a role, but according to Lafferty and Page (1997) habitat loss and degradation resulting from human disturbance is likely to have contributed to their decline. Converting coastal wetlands to marinas, highway and roadway construction, freshwater diversions, grazing, breaching of coastal lagoons, and flood control practices may have contributed (Lafferty and Page 1997).

According to Spinks et al (2003), habitat destruction, human disturbance, irrigation and exotic predators are all responsible for increasing mortality of native *Actinemys marmorata* turtle populations in California. Hence, it is difficult to separate the effects of these impacts from the effects of competition with the introduced turtle *T. scripta*.

(ii) Impacts of other introduced animals

Although the introduced bullfrog *R. catesbeiana* has been blamed for amphibian declines in much of western North America, alternative or additional causes may include introduced predatory game fishes and crayfishes (Hammerson 1982, Rosen and Schwalbe 1995). Native leopard frogs are declining in some areas where *R. catesbeiana* is absent, indicating other factors are involved (Hammerson 1982). Adams et al (2003) found that invasion by the introduced bullfrog *R. catesbeiana* in western North America is facilitated by the presence of a co-evolved non-native fish, which increases bullfrog tadpole survival by reducing predatory macroinvertebrate densities.

Luiselli et al (1997) suggest the impacts of the introduced red-eared slider *T. scripta elegans* on the native pond turtle *Emys orbicularis* in Europe may be confounded by the presence of other introduced pond turtles including *Mauremys cascipa* and *M. leprosa.*

According to Wilson and Porras (1983), the impacts of exotic fish and invertebrates have been incorrectly attributed to other exotic taxa such as reptiles and amphibians.

The introduction of the curious skink *Carlia ailanpalai* to the Mariana Islands (Guam) coincided with decline in populations of the Pacific blue-tailed skink *Emoia caeruleocauda,* and the possible eradication of the Marianas blue-tailed skink *E. atrocostata* and the mottled snake-eyed skink *Cryptoblepharus poecilopleurus* in the following decades (Lever 2003). However, the Asian musk shrew *Suncus murinus* was introduced at the same time as *C. ailanpalai* and may have displaced these native skink species in the Marianas through interspecific competition, predation or a combination of factors.

The introduced African clawed frog *X. laevis* was found to have native tidewater gobies *Eucyclogobius newberryi* in their stomachs in brackish streams and estuaries in California. Tidewater gobies have declined and predation by *X. laevis* may have played a role, but according to Lafferty and Page (1997), predation by exotic predatory fish, including yellowfin goby *Acanthogobius flavimanus*, green sunfish *Lepomis cyanellus* and rainwater killifish *Lucania parva*, may also have contributed.

(iii) Introduced diseases

Lever (2003) suggests that the decline of native vertebrate species on Guam, usually blamed on the introduced curious skink *C. ailanpalai* and introduced brown tree snake *B. irregularis*, may have been in part due to introduced diseases. However, this is speculation and is not supported by any evidence.

(iv) Climate change

Although the introduced bullfrog *R. catesbeiana* has been blamed for the decline of the California red-legged frog *R. aurora draytonii,* according to Davidson et al (2001, 2002), possible alternative causes of native frog decline include pesticide drift, changes in climate and ultraviolet-B radiation.

These confounding factors may be cumulative or may interact synergistically, such that the impact of several factors acting together is greater than the sum of the individual factors acting alone. For example, some native species might survive predation by an introduced reptile or amphibian unless habitat disturbance destroys the plants they use for shelter, so they are unable to hide. Such interactions can make it difficult to accurately understand total causes leading to specific impacts.

3.7.2 Sleepers, adaptation and niche changes

One factor which brings uncertainty to predicting impacts of introduced reptiles and amphibians is that a newly introduced exotic species may adopt a niche that differs completely from that in its native range.

When exotic species establish, they may undergo rapid evolutionary divergence in novel environments. Campbell and Echternacht (2003) took brown anoles (*Anolis sagrei*) from a single Florida population and released them on two ecologically different (forested and non-forested) islands in central Florida. The anoles adapted to the new habitats and developed significant differences in body size, population density and survival rates. Brown anoles are generally much larger where they have been introduced on mainlands compared to their size on their native Caribbean islands, indicating character release may have occurred. An alternative explanation is that food resources may be more abundant in the areas of introduction.

Introduced species can rapidly adapt to local conditions, and such rapid evolution renders them 'moving targets' for management with respect to their biotic interactions and effects on native communities (Mooney and Cleland 2001, Campbell and Echternacht 2003). These changes can include short-term, non-genetic (plastic) phenotypic adjustments and long-term evolutionary changes. Examples include character release and character displacement and the myriad effects these changes have on species interactions and community dynamics, but studies of such effects are rare (Campbell and Echternacht 2003). Body size may be influenced by abiotic factors, resource availability, population density and biotic interactions. It may also change over time and space. The outcomes of interactions between species, such as predation and competition, are likely to be affected by body size. Thus, an exotic species that changes in size due to character release will have a different effect on native biota than would be predicted from data collected from that species in its native habitats (Campbell and Echternacht 2003).

Losos et al (1997) introduced populations of *A. sagrei* anoles onto 14 small islands in the Bahamas that did not naturally contain lizards. These populations differentiated from each other in limb length and body mass over a 10–14 year period. The more the recipient island's vegetation differed from the vegetation where the lizards were sourced, the greater the magnitude of the differentiation was.

Some exotic species spread quickly. Other species may have a long lag period, but then spread may be triggered by some event such as habitat alteration, changed land use or the arrival of another exotic species (Shine et al 2000).

Although many exotic species initially establish in human-disturbed areas and may stay restricted in their distribution for decades, some may later spread to undisturbed areas of natural vegetation. Such exotic species are often called 'sleepers'. For example, Hutchinson (2001) found the Asian house gecko *H. frenatus* spent over a century in Australia confined to a few local footholds largely commensal with human settlements. It has since spread widely in a few decades and may still be expanding.

According to Butterfield et al (1997), 36 species of exotic amphibians and reptiles have established in Florida (four anurans, 28 lizards, two snakes, one turtle and one crocodile), and 22 of these species have not dispersed far beyond their sites of arrival. In some cases this may be due to insufficient time. In other cases, geographical barriers (such as being on an island) have restricted spread. Other species have had adequate time to spread but have failed to do so. Five species have undergone limited range expansion. The remaining nine species have wide continuous distributions; eight having expanded their ranges in close association with human movements. The ninth species, *Eleutherodactylus planirostris*, may be less dependent on humans and now occurs in natural habitats as well as human-occupied areas (Butterfield et al 1997).

Delays in spread and changes to niche mean that it can be decades before an exotic species starts causing harm. By the time the potential for harm is recognised, the opportunity for eradication will most often have been missed.

3.8 Adverse impacts and their significance for assessing pest status

A review of the literature on exotic reptile and amphibian introductions indicates a variety of adverse impacts may occur. These impacts include competition for resources, predation, and habitat and ecological community impacts. They are briefly described below, together with examples and their significance for risk assessments.

3.8.1 Competition for resources

Competition between introduced and native species can lead to reduced growth rates, survival and recruitment (Boland 2004, Cole et al 2005). But it is relatively difficult to demonstrate unequivocally in invaded communities (Vitousek et al 1987, Ebenhard 1988, Simberloff 1997). Competition may either be direct (interference competition) or indirect (depletion of shared resources). In interference competition, access to a resource is limited by, for example, aggressive behaviour or the release of toxins. In exploitation competition, competitors differ in their ability to exploit resources.

(i) Anurans

Various species — Growth inhibitors

When tadpoles are kept at unnaturally high densities in the laboratory, there is some evidence for interference competition between tadpoles of different species, involving growth inhibitors released into the water. Tadpoles in aquaria had inhibited growth when raised in water previously crowded by other larger tadpoles (Licht 1967). Seventeen anuran species were tested and there was no decline in this inhibition with increasing phylogenetic distance. Only *Bufo woodhousei* tadpoles seemed immune to the inhibitory effects (Licht 1967). Petranka (1989) collected water from ponds with high natural densities of tadpoles and checked to see if it inhibited growth of tadpoles in the laboratory. Growth was inhibited in only two of 13 assays and the magnitude of the inhibition was much less than for laboratory experiments with crowded tadpoles. Petranka (1989) concluded interference competition involving growth inhibitors could occur, but it is uncommon in natural tadpole assemblages. Hence, chemically-induced growth inhibition appears unlikely to be a significant impact of exotic anurans unless they reach unusually high densities.

American bullfrog (Rana catesbeiana)

There is strong field observational and experimental evidence that the bullfrog *R. catesbeiana,* introduced to western United States from the eastern states, competes for resources with native ranid frogs such as *R. pretiosa, R. pipiens, R. draytonii, R. aurora* and *R. boylii* (Moyle 1973, Bury and Whelan 1984, Fisher and Shaffer 1996, Beller 1997, Kupferberg 1997, Kiesecker and Blaustein 1998, Lawler et al 1999, Kiesecker et al 2001). According to Rosen and Schwalbe (1995), current trends suggest that inaction to control bullfrogs could lead to disappearance of three of five native ranid species in Arizona within a decade.

Kupferberg (1997) studied the invasion by the bullfrog into a northern California river system where bullfrogs are not native. Native yellow-legged frogs (*R. boylii*) were found to be almost an order of magnitude less abundant in reaches where bullfrogs were well established. Kupferberg (1997) conducted experiments to assess the potential role of larval competition in contributing to this displacement. In enclosures, bullfrog tadpoles caused a 48% reduction in survival of *R. boylii* and a 24% decline in their body mass at metamorphosis. Bullfrog tadpoles had smaller impacts on Pacific treefrogs (*Hyla regilla*) causing 16% reduction in metamorph mass, and having no significant effect on survival. Responses to bullfrogs in field settings were qualitatively similar to results seen in the smaller-scale experiments, with competition from large overwintering bullfrog larvae significantly decreasing survival and growth of native tadpoles. Competition from recently hatched bullfrog larvae also decreased survival of *R. boylii* and *H. regilla*. Bullfrog tadpoles also significantly affected benthic algae, although effects varied across sites. Competition appeared to be mediated by algal resources, and there was no evidence for behavioural or chemical interference. According to Kupferberg (1997), amphibian populations are strongly influenced by changes in recruitment, so native species may decline where bullfrogs invade and compete with larvae.

Lawler et al (1999) found that in the presence of bullfrog tadpoles, the survivorship of tadpoles of the California red-legged frog *R. draytonii* was reduced to 5% from 34% in artificial ponds. Bullfrogs nearly eliminated red-legged frog recruitment in this experiment. This study provides experimental evidence that bullfrogs may play a role in the decline of the California red-legged frog. The mechanism was not identified but competition was likely, although predation possibly contributed, as bullfrog tadpoles will eat red-legged frog tadpoles.

In field enclosure experiments, tadpoles of the native northern red-legged frog *R. aurora* altered their microhabitat use, in the presence of bullfrog adults and tadpoles (Kiesecker and Blaustein 1998). Growth and development was also affected, with time to metamorphosis increased and mass at metamorphosis decreased for *R. aurora* tadpoles in the presence of either tadpoles or adult bullfrogs. Survival of *R. aurora* was affected when tadpoles were exposed to both tadpole and adult bullfrogs at the same time. Adult bullfrogs decreased *R. aurora* metamorph survival by one third. When bullfrogs were combined with smallmouth bass (*Micropterus dolomieui*), another introduced species, these negative impacts were enhanced because of interactive effects. According to Kiesecker and Blaustein (1998), the mechanism is unclear but interference competition was considered to be the likely cause. However, predation possibly contributed, as bullfrog tadpoles eat tadpoles of other species, including *R. aurora* in the laboratory.

Behavioural observations by Kiesecker et al (2001) indicate that a passive interference mechanism is likely to be responsible for the outcome of interactions between bullfrogs and native red-legged frogs (*R. aurora*). Kiesecker et al (2001) found survival to metamorphosis and mass at metamorphosis were reduced when red-legged frog tadpoles were exposed to bullfrogs in clumped-resource ponds and suggest that clumped resources can intensify interspecific competition. This competition may influence the success of exotics when human-induced habitat alteration affects resource distribution. These authors conclude that understanding the context-dependent nature of interactions will be necessary if we are to predict invasion success and control the impact of exotics on natives.

According to Werner (1994), competitive effects on growth rates can have manifold effects on anuran fitness; for example, by protracting the time tadpoles are vulnerable to predators. Also, it may cause larvae to overwinter for additional seasons before metamorphosing and mortality in winter can be high.

According to Boyd (1975, cited in Lever 2003), a high density of *R. catesbeiana* tadpoles can inhibit reproduction by guppies (*Poecilia reticulate*) in the laboratory.

The possible impacts of adult terrestrial bullfrogs as competitors are considerable but difficult to quantify. Morey and Guinn (1987) found a high degree of diet overlap of arthropod taxa between juvenile terrestrial bullfrogs dispersing around vernal pools in California and adult native frogs breeding there. It is not known though whether competition for insect resources limit native frog populations.

Common frog (Rana temporaria)

Griffiths (1991) conducted a replicated pond experiment and showed that high densities of *R. temporaria* tadpoles resulted in slower growth, smaller size at metamorphosis, prolonged development and reduced survival of natterjack toad *B. calamita* tadpoles. Both *B. calamita* and *R. temporaria* are native species in United Kingdom but *B. calamita* is confined to inland heath and coastal dune systems and degradation has resulted in incursions of *R. temporaria* into *B. calamita* habitats.

Cane toad (Bufo marinus)

According to Freeland (1984) and Freeland and Martin (1985), perceived competitive effects from introduced cane toads on native fauna include: adults competing for food with native fauna, adults outcompeting native fauna for shelter and resting places, and tadpoles competing with native amphibians in breeding habitat. Much evidence of the impacts of the cane toad in Australia is anecdotal with little data to support the claims of negative impacts on native fauna, or to refute them (Freeland 1987, Crossland 1998, Catling et al 1999). However, cane toads are extremely aggressive in laboratory tests when competing for food with *B. americanus.* Freeland (1984) reported anecdotal evidence from New Guinea that native geckos and skinks that sheltered under logs and rocks declined after cane toads arrived, although the mechanism was unknown. Freeland (1984) also suggested that because cane toads are highly fecund and their tadpoles collect in large aggregations, this may confer competitive superiority over native Australian frogs such as *Litoria caerulea*. According to Crossland (1997) introduced *B. marinus* tadpoles may compete with native aquatic fauna in northern Queensland.

Williamson (1999) reported preliminary findings of competition trials between *B. marinus* tadpoles and native anurans in Australia. The trials were conducted in small artificial ponds. The results indicated that *B. marinus* reduced the growth of three native frog species (*Limnodynastes tasmaniensis*, *L. terraereginae* and *Notaden bennetti*), and in some trials reduced the survival of two species (*L. tasmaniensis* and *L. terraereginae*). One of two trials conducted in small enclosures in a permanent water body indicated that *B. marinus* had a negative effect on growth of *L. tasmaniensis*. A survey of 30 breeding sites in the area found that *B. marinus* used only a small number of water bodies in one breeding season and showed little overlap of pool use with most native species. Therefore, although *B. marinus* may negatively affect growth and survival of native anurans under some circumstances, the impact of *B. marinus* may be minimal if there are always many breeding sites where native anurans can breed in the absence of *B. marinus*.

According to Catling et al (1999), where *B. marinus* is expanding in the Northern Territory of Australia, small reptile fauna (and especially small skinks) may decline in diversity and abundance over the long term due to indirect competition, because the toads deplete their invertebrate food supply. Catling et al (1999) assessed the effects of expanding populations of *B. marinus* in the territory and found that cane toads significantly depleted the abundance of insects (Coleoptera), so could potentially lead to competition for food with native insectivores.

Freeland and Kerin (1988) demonstrated that *B. marinus* does not substantially overlap in resource use with four species of native frogs in Australia. Similarly, Williamson (1999) noted that native frogs and *B. marinus* rarely use the same breeding ponds under natural conditions and concluded that this minimised the potential for cane toads to have competitive impacts. In an empirical study conducted on the edge of the cane toad's invasion pathway, Catling et al (1999) found no evidence of a direct long-term effect of cane toads on native amphibian abundance or diversity in northern Australia. Boland (2004) suggested introduced *B. marinus* has the potential to cause a significant impact on a wide array of native fauna through competition for shelter sites and even raised the possibility that cane toads might evict native animals from their burrows. However the potential role of *B. marinus* as a competitor with native fauna for shelter sites has not been investigated. The exception is the study by Boland (2004), which showed cane toads evict nesting rainbow bee-eaters (*Merops ornatus*) from their nest burrows. Chicks that were too large to be eaten by the cane toads usually starved because parent birds were unable to reach them, due to cane toads occupying the nest tunnel.

According to King (1968) *B. marinus* is replacing the native southern toad *B. terrestris* in the cities of southern Florida. Where *B. marinus* and native *B. terrestris* populations overlap, the transformation times of the larvae of *B. terrestris* are abbreviated while those of *B. marinus* are lengthened (Rossi 1981). Bartlett and Bartlett (1999) suggest such competition may contribute to scarcity of *B. terrestris* in some places. According to Rabor (1952) and Alcala (1957), introduced *B. marinus* in the Philippines occur mainly on open disturbed land where sympatric native species (mainly *Kaloula picta*, *Rana cancrivora*, *Rana vittigera* and *Polypedates leucomystax*) remain abundant.

Cuban treefrog (Osteopilus septentrionalis)

Declines of some native anurans, such as *Hyla cinerea* and *H. squirella,* in south Florida have been reported and these declines are anecdotally correlated with the arrival of the exotic Cuban treefrog (*O. septentrionalis*). Competition has been suggested as a mechanism (Crockett et al 2002) but the declines could also be attributable to predation by adult exotic anurans or to concurrent effects of habitat destruction (Smith 2005). The ability of *O. septentrionalis* to disperse and to penetrate relatively undisturbed habitats suggests that future adverse impacts on native anurans are possible (Smith 2005).

Smith (2005) used laboratory manipulations to examine the competitive effects of the larvae of two introduced anurans (the cane toad *B. marinus* and the Cuban treefrog *O. septentrionalis*) on the growth and development of the larvae of two anurans native to Florida (the southern toad *Bufo terrestris* and the green treefrog *Hyla cinerea*). The presence of *O. septentrionalis* larvae consistently reduced growth rates and delayed development and metamorphosis of tadpoles of both native species and *B. terrestris* had a smaller mass at metamorphosis. *H. cinerea* tadpoles transformed at greater body masses when reared with the rapidly transforming exotic species as a result of competitive release. The negative effects of *O. septentrionalis* on native tadpoles were generally significant whether the tadpoles were exposed to *O. septentrionalis* alone or in combination with *B. marinus*. Neither exotic species significantly decreased the survival of native tadpoles, although a trend toward decreased survival was evident for *H. cinerea*. These results suggest that exotic tadpoles may adversely affect native tadpole communities as a result of interspecific competition. Competition is an important ecological factor in tadpole communities and there is a significant potential for competition between tadpoles of native and exotic species.

Coqui (Eleutherodactylus coqui)

Kraus et al (1999) suggested one possible impact of the *E. coqui* frogs in Hawaii is competition with Hawaiian birds for insect prey.

Piping frog (Eleutherodactylus johnstonei)

Kaiser et al (1994) suggested introduced *E. johnstonei* in Grenada (West Indies) may have led to the decline of the native *E. euphronides* through interspecific competition.

African clawed frog (Xenopus laevis)

Lobos and Jaksic (2005) suggested *X. laevis* in Chile may be competing with native anurans.

(ii) Reptiles

Based on studies of introduced exotics on Pacific islands and manipulative experiments, Case and Bolger (1991ab) presented evidence supporting the hypothesis that predation and competition set important constraints on the distribution, colonisation (establishment) and abundance of reptiles (predominantly lizards) on islands. They suggested competition from introduced exotics has led to changes in abundance of native species, but also considered competition is unlikely to lead to extinctions of reptile populations.

According to Wilson and Porras (1983) most exotic reptiles and amphibians introduced to south Florida are primarily restricted in distribution to urban areas were few native reptiles and amphibians occur, and only two native lizards appear to be abundant. Wilson and

Porras (1983) considered there is thus little opportunity for competition between native and introduced lizards.

Thermal conditions have been directly related to fitness in reptiles and thermally appropriate basking sites can be a limited resource over which competition may occur in lizards (Melville 2002).

Red-eared slider (Trachemys scripta)

According to Cadi and Joly (2004), *T. scripta* has established exotic breeding populations in Italy, Spain and southern France. The exotic *T. scripta* may be ecologically dominant over the native pond turtle *Emys obicularis* (an endangered species in Europe) and compete with *E. obicularis* for resources (Luiselli et al 1997). The outcome of competition depends on differences in the respective abilities of native and exotic species to use habitat resources.

Cadi and Joly (2003) used experimental ponds to show *E. obicularis* shifted their basking activity to lower quality sites while *T. scripta* occupied the better sites — suggesting *T. scripta* had dominance. Basking is important for turtles because their metabolism is governed by body temperature. Cadi and Joly (2004) constructed four ponds, each 240 square metres, with natural food and vegetation. In two of the ponds, eight individuals of each species were introduced, matched for body size and with a balanced sex ratio. In the other two ponds, only eight *E. obicularis* turtles were introduced. *E. obicularis* lost weight in the mixed ponds but *T. scripta* did not. The body weights of *E. obicularis* were stable in the single species ponds. Mortality in *E. obicularis* was also significantly higher in the mixed species ponds. In contrast, *T. scripta* had high survival and growth. Cadi and Joly (2004) suggested *T. scripta* can be expected to have a competitive advantage over native *E. obicularis* because of the slider's lower age at maturity, higher fecundity and larger adult body size. These authors suggested the two species may compete for food, nesting sites and basking places and could be involved in interference competition. Their experiment demonstrates competitive dominance by *T. scripta* over *E. obicularis*, but density was higher than for wild populations.

Field observations by Spinks et al (2003) in an urban Californian site suggested competition for basking sites may exist between introduced *T. scripta* and the native pond turtle *Actinemys marmorata*. In this study, basking sites were limited because much of the water/shore interface was concrete or wire-wrapped rock, so turtles of all species were usually observed basking at a few prime sites. At these sites, interspecific confrontations were frequently observed. In some instances, as *A. marmorata* approached occupied basking sites, they gaped at basking *T. scripta.* Lindeman (1999) has shown that in confrontations for basking sites between *T. scripta* and other emydid turtles, the largest turtle successfully displaces the smaller, regardless of species. Female *T. scripta* can grow to more than twice the size of *A. marmorata*, and Spinks et al (2003) found *T. scripta* weighed, on average, 38% more than *A. marmorata*. If the outcome of competitive interactions at basking sites is determined by size, then it is likely that *T. scripta* will out-compete *A. marmorata* for basking sites. Spinks et al (2003) concluded that further observations are needed to determine the extent to which *A. marmorata* may be negatively affected.

It has also been suggested that *T. scripta* might compete for food and basking and nesting sites with native turtles in France (*Mauremys caspica,* Lever 2003), Israel (*M. caspica,* Bouskila 1986), and South Africa (*Pelomedusa subrufa,* Newberry 1984).

Common house gecko (Hemidactylus frenatus)

A native, unisexual gecko (*Lepidodactylus lugubris*) declines numerically when the sexual gecko *H. frenatus* invades urban/suburban habitats throughout the Pacific (Petren and Case 1996). Competitive displacement occurs rapidly, facilitated by clumped insect resources. The two species show nearly complete diet overlap and insects are a limiting resource. *H. frenatus* depletes insect resources to lower levels than *L. lugubris,* which results in reduced rates of resource acquisition by *L. lugubris*. This reduced resource acquisition translates into significant reductions in body condition, fecundity and survivorship of *L. lugubris* individuals. Increasing *L. lugubris* density has negligible effect on *H. frenatus*. The superior harvesting ability of *H. frenatus* is most pronounced when insects are clumped spatially and temporarily, and is attributable to a variety of species-specific traits such as their larger body size, faster running speed, and reduced intraspecific interference while foraging. Petren and Case (1996) concluded that clumped resources (eg around artificial lights) can increase interspecific exploitation competition, and this mechanism may contribute to species turnover when human environmental alterations redistribute resources. Petren and Case (1996) rarely observed interference competition in the form of active, directed agonistic attacks and both species shared shelters during the day, often at high densities. This conflicts with the findings of Brown et al (2002) who found that *L. lugubris* avoided sharing hiding places with *H. frenatus*.

The introduced *H. frenatus* was first found in Hawaii in 1951, and has competitively displaced the mourning gecko (*L. lugubris*), and possibly also the fox or Polynesian gecko (*H. garnotii*) and the stump-toed gecko (*Gehyra mutilata*) on buildings (Case et al 1994). *L. lugubris* is still abundant, often in association with *H. frenatus* in shoreline vegetation, but *L. lugubris* declines on buildings when *H. frenatus* is present. All three displaced species were introduced to Hawaii by Polynesian travellers about 400 AD, and all became scarce or declined in abundance in urban/suburban habitats when house gecko numbers increased. *L. lugubris* is nearly eight times more abundant in urban/suburban habitats on Pacific islands where the house gecko is absent, than in such habitats on islands where *H. frenatus* is present (Case et al 1994). Experimental evidence supports a role for competitive displacement for feeding sites on walls near electric lights where prey insects congregate, with the larger house gecko aggressively defending feeding patches against mourning geckos. Case et al (1994) suggested in more complex forest habitats, where food resources are not so aggregated, such aggressive displacement may not occur.

Brown et al (2002) conducted experiments to see if factors other than exploitative competition for food could contribute to observed declines in established populations of *L. lugubris* around artificial lights when *H. frenatus* invades an environment. Brown et al (2002) found *L. lugubris* avoided sharing hiding places with *H. frenatus,* which made them more vulnerable to predators. *L. lugubris* also laid more eggs when housed with another *L. lugubris* than when housed with an *H. frenatus.* Also, *L. lugubris* housed in enclosures previously occupied by *H. frenatus* required more time for egg development and laying than *L. lugubris* housed in enclosures previously occupied by *L. lugubris*. This finding suggests *L. lugubris* fecundity may be negatively affected by exudates from *H. frenatus.*

Cole et al (2005) investigated the potential impacts of the exotic house gecko *H. frenatus* on endemic geckos in non-developed relatively undisturbed areas in the Mascarene Islands. These authors found spatial segregation occurs between introduced *H. frenatus* and endemic night geckos (*Nactus coindemirensis, N. durrelli* and *N. serpensinsula)* throughout the Mascarene Islands. All three species of the night gecko are smaller or of similar size to the house gecko and sub-fossil remains reveal that the night geckos have undergone a catastrophic reduction in range (Cole 2002). Cole et al (2005) present evidence that the introduced house gecko has caused the catastrophic decline and extinction of the endemic night gecko populations. Neither habitat destruction nor any other introduced competitor or predator can account for the fragmentation of the night geckos' population as accurately as the distribution of the house gecko.

Cole et al (2005) tested competition for enemy-free space in experimental enclosures and showed that *H. frenatus* displaces the endemic *N. coindemirensis* and *N. durrelli* from favoured positions close to and from refugia. This displacement increases the risk of exposure of the endemic geckos to stochastic events, such as cyclones and predation from introduced predators such as brown rats (*Rattus norvegicus)*, ship rats (*R. rattus*), cats (*Felis catus*) and musk shrews (*Suncus murinus*). Cole et al (2005) suggested that in addition to these mammalian predators, in the presence of *H. frenatus,* some avian and reptilian predators may also have had a significant role in determining the current distribution of the night geckos due to their exclusion from refugia.

Interactions between *H. frenatus* and both *N. coindemirensis* and *N. durrelli* were mostly aggressive, with the introduced gecko frequently observed stalking, lunging towards and biting the endemics. For example, two individual *N. coindemirensis* lost toes, a further two individuals lost their tails and one male was eaten. The loss of toes and tails has been shown to reduce locomotion and gripping ability: tail loss decreases growth, reduces fecundity, reduces home-range size and enhances loss of territories in other lizard species. Furthermore, tail regeneration in females of some gecko species can inhibit reproduction. Therefore, in addition to the likely increased mortality risk arising from exclusion from refugia, the injuries sustained by night geckos through direct aggressive interactions with *H. frenatus* were likely to have a further direct impact upon the survival and reproductive success of individuals, especially the smaller *N. coindemirensis*. These findings by Cole et al (2005) support the hypothesis that *H. frenatus* led to the fragmentation and extirpation of endemic *Nactus* populations. The findings also demonstrate that in experimental enclosures, asymmetrical aggressive interactions are responsible for the competitive exclusion of both *N. coindemirensis* and *N. durrelli* from daytime refugia by *H. frenatus*, such that individuals of native species were forced to occupy areas approximately twice as far from refugia in the presence of *H. frenatus* versus its absence.

Cole (personal communication, University of Bristol, 2005) has also found evidence that *H. frenatus* in the Mascarene Islands is having a negative impact on the endemic populations of ornate day gecko *Phelsuma ornata* through indirect competitive interactions for food resources, and increased susceptibility to parasites. These interactions are entirely asymmetrical, whereby no detectable negative effects are experienced by *H. frenatus*.

Italian wall lizard (Podarcis sicula)

The lacertid lizard *P. sicula* has spread and replaced the native wall lizard *P. melisellensis* throughout coastal areas and numerous islands in the Mediterranean (Nevo et al 1972). Following experimental introductions of *P. sicula* to islands inhabited by *P. melisellensis,* it was suggested that the former species were competitively excluding the natives (Radovanovic 1965, cited in Cole et al 2005). The causal mechanism of this exclusion has been demonstrated using experimental enclosures to show that juvenile *P. sicula* outcompete juvenile *P. melisellensis* for microhabitats of preferred thermal properties, through asymmetric aggressive interactions, thus affecting growth and fitness of *P. melisellensis* (Downes and Bauwens 2004).

According to Capula (1993, 1994), *P. sicula* in the Aeolian Islands (in the Mediterranean) has reduced the range and eradicated many populations of the native wall lizard *P. raffonei* partly through competitive exclusion.

Anoles (Anolis spp)

Anolis lizards have been widely introduced, usually unintentionally, throughout the Caribbean, Florida and elsewhere. Experiments with anoles demonstrate competition for resources such as prey and perch sites (Pacala and Roughgarden 1982).

The introduced brown anole *A. sagrei* competes with native lizards (Campbell 2000, Gerber and Echternacht 2000, Vincent 2002, Campbell and Echternacht 2003). *A. sagrei* is expanding its range in Grand Cayman in the Caribbean and is now more common in some habitats than the native anole *A. conspersus.* According to Losos et al (1993), competition may be occurring between the two species. Comparisons with studies prior to the arrival of *A. sagrei* indicate that in open habitats where *A. sagrei* is now abundant, *A. conspersus* perches higher, but in closed habitats where *A. sagrei* is absent, no change in perch height is evident. Losos and Spiller (1999) demonstrated competition between *A. sagrei* and *A. carolinensis.* These authors released propagules of five individuals (three females, mostly gravid and two males) of *A. sagrei* on ten very small islands in the Bahamas. The *A. sagrei* populations thrived on nine of the ten islands. In contrast, when five individuals of *A. carolinensis* were introduced to ten islands, many became extinct within three years. On the five islands where both species were introduced, populations of *A. carolinensis* were smaller and individuals tended to perch higher than they did on islands where *A. sagrei* was absent. Conversely the presence of *A. carolinensis* had little long-term impact on *A. sagrei* populations, although in the initial year following introduction *A. sagrei* populations were five times higher on islands without *A. carolinensis* than on islands with this species. But once *A. carolinensis* numbers declined on sympatric islands, the numbers of *A. sagrei* increased to match the numbers of *A. sagrei* on allopatric islands.

Campbell (1999/2000) investigated interactions between introduced *A. sagrei* and native *A. carolinensis* in Florida. Where the two species occurred together, *A. carolinensis* shifted their perch height upwards and were excluded from several habitats, presumably by aggression from *A. sagrei* below. At the higher perch levels, dietary prey species were less diverse and abundant. Campbell (1999/2000) found that where both species occurred, *A. sagrei* numbers increased while *A. carolinensis* numbers declined. Campbell (1999/2000) concluded interference competition (causing shifts in perch height) and exploitative competition (causing shifts in diet) could cause the declines in numbers of the native species. Campbell (1999/2000) also reported that predation by the vastly more numerous *A. sagrei* adults on juvenile *A. carolinensis* contributed to the decline of the latter, but suggested that where dense shrub cover exists the two species should be able to co-exist.

Fitch et al (1989) conducted a field study of *A. cristatellus* (native to Puerto Rico) introduced in the Dominican Republic and found the introduced species displaces two native *Anolis* species, *A. chlorocyanus* and *A. cybotes*, by competition and/or predation.

King (1966) suggested competition occurs between introduced *A. distichus* and native *A. carolinensis* in southern Florida. King (1968) suggested competition is causing the native *A. carolinensis* in southern Florida to be replaced by the introduced *A. distichus* and *A. sagrei*. Losos (1996) suggested that introduced *A. extremus* competes with introduced *A. grahami* in Bermuda for habitat and food and slows the latter's rate of spread, but the two species can co-exist.

Eastern grass skink (Lampropholis delicata)

West (1979) suggested that introduced *L. delicata* in New Zealand might compete for food with the native copper skink (*Cyclodina aenea*), particularly because the introduced species reaches very high densities.

Risk assessment significance: Competition by exotic reptiles and amphibians has the potential to be highly detrimental to native species. However, scientific knowledge is sparse and currently inadequate to allow reliable predictions about which exotic species will have the worst impacts when they are introduced to new environments.

3.8.2 Predation

According to Freeland (1984), predation impact is likely to depend on:

- predator population density and dynamics in prey habitats
- rates and patterns of prey consumptions, as determined by:
  - relative availability of different prey species
  - spatial and age distributions of predator populations
- capacity of individual predators to increase prey consumption with increasing prey density (functional response)
- capacity of predator populations to increase as prey populations increase (numerical response).

Unfortunately, these factors have rarely been studied for exotic reptile and amphibian predator species and their prey populations.

Stomach content analyses of exotic species usually reveal little about the potential significance of exotic species as predators of native fauna. This is because one species' predation on another species may not result in reducing the population density of the prey species. Even if there is a population effect, it will often be difficult to assess the impact of exotic reptiles and amphibians as predators. This is because there may be only a brief window of time in which sensitive native species have high enough relative abundances to be detected in a diet study (Kupferberg 1997).

(i) Anurans

Adult anuran amphibians generally rely on invertebrates for most of their diet, but may prey on other vertebrates. Although primarily herbivorous, many tadpoles also prey on the eggs, hatchlings or tadpoles of other anurans (Crossland 1998). The ability of many tadpoles to facultatively shift from an herbivorous to a predatory diet means that they may play an important role in structuring aquatic systems (Crossland 1998).

American bullfrog (Rana catesbeiana)

Originally native to eastern North America, *R. catesbeiana* has been widely introduced in the western United States. In these western states, the bullfrog's enemies (basses, pikes, snapping turtles and water snakes) are absent and *R. catesbeiana* attains high population densities (eg Rosen and Schwalbe 1995). *R. catesbeiana* tadpoles are strongly herbivorous, mainly eating detritus and algae. However, in the laboratory, bullfrog tadpoles eat the eggs and tadpoles of the native frog *R. blairi* (Bury and Whelan 1984). Adult bullfrogs are carnivores, eating any animal smaller than themselves, mainly crustaceans and insects, but also rodents, bats, frogs, birds, fish and reptiles (Bury and Whelan, 1984, Rosen and Schwalbe 1995). Out of 252 stomach contents examined by Schwalbe and Rosen (1988), 14.6% contained vertebrates and the dominant vertebrate found was other anurans, suggesting that predation may be significant for native frogs.

Where *R. catesbeiana* has been introduced in the western United States, its predatory habits have implicated it in the decline of native ranid frogs (*R. pipiens, R. pretiosa, R. onca, R. boylii,* *R. aurora, R. blairi, R. fisheri, R. yavapaiensis* and *R. chiricahuensis*) and the Mexican garter snake *Thamnophis eques* (Moyle 1973, Bury and Whelan 1984, Schwalbe and Rosen 1988, Corn 1994, Rosen and Schwalbe 1995, Beller 1997, Hecnar and M'Closkey 1997, Kupferberg 1997). In Arizona, Schwalbe and Rosen (1988) found only one site out of 80 where *R. yavapaiensis* and *R. chiricahuensis* coexist with *R. catesbeiana*. Bullfrogs are also suspected to be significant predators of hatchling and juvenile western pond turtles *(Actinemys marmorata,* Milner 1986). Bury and Whelan (1984) reported that *R. catesbeiana* bullfrogs ate all the Pacific treefrogs (*Hyla regilla*) from a mill pond and had generally reduced this species in Oregon. Similarly in Italy, farmers accuse *R. catesbeiana* of preying on native ranid species including *R. temporaria, R. dalmatina, R. graeca, R. lessonae, R. esculenta* and *R. latastei* and on native fish (Lever 2003). Stomach content analyses of *R. catesbeiana* in Italy found other frogs, snakes and birds. In Spain, it has been suggested that *R. catesbeiana* could threaten the native *R. perezi* (Moyle 1973). Competition by *R. catesbeiana* and human disturbance may also have played a role in the decline of native ranid frog species in the United States and Europe (Bury and Whelan 1984).

According to Rosen and Schwalbe (1995), extensive cannibalism by *R. catesbeiana* renders them especially potent predators at the population level. The tadpoles require only perennial water and grazeable plant material. Hence, transforming young can sustain a dense adult bullfrog population even if alternate prey is depleted. This behaviour may increase the probability that native species may be extirpated by bullfrog predation.

Rosen and Schwalbe (1995) conducted a removal experiment with *R. catesbeiana*, and monitored the population structure of two native prey species: the Mexican garter snake (*Thamnophis eques*) and the Chiricahua leopard frog (*R. chiricahuensis*). Under the bullfrog-removal treatment, numerous young snakes (1–3 years old) showed successful reproduction in apparently intact populations. In contrast, the bullfrog-affected populations were composed mainly of older snakes. Once the young snakes outgrew vulnerability to bullfrog predation, they survived well. Bullfrogs ate the last of the *R. chiricahuensis* frogs at the study sites.

Kiesecker and Blaustein (1997) studied eight populations of the red-legged frog *R. aurora,* to examine responses of their tadpoles to *R. catesbeiana*, an introduced predator. These authors also assessed predation rates by *R. catesbeiana.* The *R. aurora* tadpoles were either from syntopic (ie coexisted with *R. catesbeiana*) or allotopic (ie not previously exposed to *R. catesbeiana)* populations. Syntopic *R. aurora* tadpoles significantly reduced their activity and increased their refuge use when presented with the chemical cues of both tadpoles and adult *R. catesbeiana*. In contrast, allotopic tadpoles did not significantly alter their behaviour in the presence of either *R. catesbeiana* adults or larvae. Predation by *R. catesbeiana* was lower in syntopic than in allotopic populations of *R. aurora* tadpoles. These results show syntopic *R. aurora* tadpoles avoid predation by *R. catesbeiana* more efficiently than do *R. aurora* tadpoles from allotopic populations, which appeared not to possess adaptations that would prevent a negative encounter.

Coqui (Eleutherodactylus coqui)

E. coqui frogs are introduced to Hawaii and attain extremely high densities. Natural populations of *E. coqui* can reach densities of 20,570 adults per hectare (Stewart and Rand 1991); in Hawaii, they can attain more than three times that density (Woolbright et al 2006). *E. coqui* can invade mid-elevation moist and rainforests where it can be expected to exert tremendous predation pressure on a variety of native arthropods (Kraus et al 1999). Tummons (2003) suggests dense populations of coqui frogs may eat over 200 kilograms of arthropods per hectare per year.

Cane toad (Bufo marinus)

Crossland (1998) investigated the role of *B. marinus* tadpoles as predators of Australian native anuran eggs, hatchlings and tadpoles. In controlled laboratory experiments, neither small nor large *B. marinus* tadpoles were significant predators on these early life stages of native anurans.

Boland (2004) suggested introduced *B. marinus* has the potential to cause a significant impact on a wide array of native fauna through their role as active predators. Adult cane toads mainly eat ants and beetles, but also take small birds, rats, mice, planigale marsupials (*Planigale maculate*), frogs, skinks, geckos and snakes (Boland 2004). Cane toads use both visual and olfactory cues to locate prey. In Australia, introduced *B. marinus* ruined one third of nest attempts by native ground-nesting rainbow bee-eaters (*Merops ornatus*) by usurping their nest burrows and preying on their eggs and nestlings (Boland 2004). This behaviour had a significant effect on rainbow bee-eater populations, reducing nest productivity from 1.2 fledglings per nest in the absence of *B. marinus* to 0.8 fledglings per nest where toads were present. It is also possible that predation by cane toads affects other ground-nesting native vertebrates, particularly small tunnel-nesting birds such as pardalotes and kingfishers, but this has not been investigated (Boland 2004).

B. marinus has been implicated in the decline of many native frog populations in their introduced range (Freeland 1984, Clarke et al 2001). It is believed *B. marinus* may directly prey on the eggs and young of native frog species or simply poison native tadpoles and adult frogs that attempt to consume either the eggs or tadpoles of the cane toads. As yet there is little substantial evidence to confirm these claims (Crossland 1998, Crossland and Alford 1998). A study by Crossland (1998) found that young cane toads were not significant predators of either the eggs or tadpoles of native amphibian species. Catling et al (1999) found few short-term effects in the diversity and abundance of native mammals and reptiles after the initial invasion of cane toads into areas of northern Australia. In Florida, where introduced *B. marinus* and native Southern toad (*B. terrestris*) ranges overlap, *B. marinus* preys on *B. terrestris,* and Rossi (1981) suggested such predation may contribute to scarcity of the Southern toad in some places.

Although the diet of introduced *B. marinus* is primarily composed of arthropods, few attempts have been made to quantify the impacts of cane toads on invertebrate communities (Freeland and Martin 1985, Clarke et al 2001). Catling et al (1999) found there were some short-term negative effects to coleopteran populations in northern Australia after the invasion of cane toads. These were the result of direct predation of beetles in the areas of initial invasion. According to Lever (2003), introduced *B. marinus* in Japan preys on and has had adverse impacts on native terrestrial fauna, particularly snails and insects.

Cuban treefrog (Osteopilus septentrionalis)

The Cuban tree frog preys on native Southern toads (*Bufo terrestris*), Eastern narrow-mouth toads (*Gastrophryne carolinensis*), Southern leopard toads (*Rana sphenocephala*) and treefrogs (*Hyla cinerea, H. v. versicolor* and *H. squirella* and conspecifics) in Florida (King 1968, Crockett et al 2002, Meshaka et al 2004, Butterfield et al 1997). According to Ashton and Ashton (1988, cited in Lever 2003), preliminary research suggests a negative association between the numbers of *O. septentrionalis* and those of *H. cinerea* and *H. squirrela*, at least partly due to predation by *O. septentrionalis*. Cuban treefrog adults are voracious predators and are also cannibalistic. Cuban treefrog tadpoles are also carnivorous and are known to eat other tadpoles (Babbitt and Meshaka 2000). Wilson and Porras (1983) suggested *O. septentrionalis* has 'great potential' to displace native frogs in southern Florida. However, despite circumstantial evidence, no study has shown that *O. septentrionalis* reduces populations of native frogs in natural areas.

African clawed frog (Xenopus laevis)

X. laevis frogs are mainly aquatic and reach densities up to 8.9 frogs per square metre in some locations (Measey and Tinsley 1998, Lobos and Measey 2002). There are concerns about predation impacts of introduced *X. laevis* in the United Kingdom, the United States and Chile (Lafferty and Page 1997, Tyler 2001, Lobos and Measey 2002, Lever 2003, Lobos and Jaksic 2005). The diet of *X. laevis* in both native and non-native habitats is mainly invertebrates, although small vertebrates (fish, amphibians and terrestrial vertebrates) have also been found in their diet (Lafferty and Page 1997, Measey 1998bc, Lobos and Measey 2002, Lobos and Jaksic 2005).

In brackish streams and estuaries in California, *X. laevis* were found to have eaten native tidewater gobies *Eucyclogobius newberryi* (Lafferty and Page 1997). Tidewater gobies have declined there, but other factors are also likely to have contributed. According to Lafferty and Page (1997) *X. laevis* can prey on vulnerable finfish and could threaten the survival of the tidewater goby *E. newberryi* in Santa Clara River, California. Tinsley and McCoid (1996) suggested predation by *X. laevis* might threaten survival of the endangered unarmoured threespine stickleback (*Gasterosteus aculeatus williamsoni*) in Placerita Canyon, California. *X. laevis* may also threaten native North American amphibians, such as the Western toad *Bufo boreas* and tree frogs such as *Hyla californiae* (Lever 2003).

In Chile, introduced *X. laevis* invades pristine habitats and reaches densities up to 0.25 frogs per square metre. According to Lobos and Measey (2002), at such high densities predation is likely to have a significant impact on prey populations. Potential predation by *X. laevis* on eggs, larvae and metamorphs of endangered or vulnerable native amphibians is a cause for concern, although no studies have yet found evidence for impacts on native prey populations in Chile.

(ii) Reptiles

Brown treesnake (Boiga irregularis)

The arrival and proliferation of brown treesnakes on Guam in the Mariana Islands led to the loss of most of the island's indigenous forest vertebrates through predation by the snake (Savidge 1987; Fritts and Rodda 1995, 1998; Rodda et al 1999; Amand 2000; Wiles et al 2003). *B. irregularis* is able to feed on almost any small vertebrate it encounters due to its wide size range. The snake's nocturnal and arboreal habits make roosting and nesting birds, eggs and nestlings all vulnerable to predation. Following introduction to Guam, snakes irrupted to high densities, up to 80–120 snakes per hectare in one dense population at the peak of the irruption (Rodda et al 1999). By including abundant small reptiles in its diet, *B. irregularis* maintained high densities in forest and second growth habitat while exterminating more vulnerable prey. Lever (2003) suggested the brown tree snake's ability (in common with other reptiles) to go for long periods without feeding enables it to continue as an effective predator even if prey abundance fluctuates. On Guam, this snake extirpated nine native bird species, and was probably a primary cause of the extirpations of five native lizard species and two bat species, which has meant the extinction of these species in many cases (Savidge 1987; Rodda and Fritts 1992; Rodda et al 1997, 1999; Fritts and Rodda 1998; Amand 2000). Predation by *B. irregularis* led to serious reduction of most of the island's remaining 16 resident bird species (Wiles et al 2003). Initially, native birds were an important food item for the introduced treesnake, but they became scarce and were no longer a major part of the snake's diet (Rodda et al 1999). Once the prey populations declined, snake populations also declined, but episodic high snake densities may still occur. Rodda et al (1999) estimated that a dense population of *B. irregularis* on Guam has the capacity to annually consume about 18–30 times the biomass of adult native birds that used to be present under the most favourable conditions. By 1980, most forested areas on Guam retained only three native vertebrates, all of which were small lizards (Fritts and Rodda 1998).

Wiles et al (2003) analysed two sets of survey data gathered in northern Guam between 1976 and 1998 and reviewed unpublished sources to provide a comprehensive account of the impact of brown tree snakes on the island's birds. Their results indicate that 22 species, including 17 of 18 native species, were severely affected by snakes. Twelve species were likely extirpated as breeding residents on the main island, eight others experienced declines of 90% throughout the island (or at least in the north), and two were kept at reduced population levels during all or much of the study. Declines of 90% occurred rapidly, averaging just 8.9 years along three roadside survey routes combined and 1.6 years at a 100 hectare forested study site.

Common house gecko (Hemidactylus frenatus)

In the laboratory, common house geckos prey on juvenile mourning geckos (*Lepidodactylus lugubris*) but the reverse is not true, and stomach analyses of wild-caught house geckos revealed few juvenile mourning geckos (Case et al 1994).

According to Cogger et al (1983), in parts of its Australian range, the introduced *H. frenatus* has displaced native *Gehyra* spp as the house gecko in settled areas. Lever (2003) suggested that on Christmas Island, the introduced *H. frenatus* has the potential to adversely affect the endemic Christmas Island gecko (*Lepidodactylus listeri*).

Petren and Case (1996) demonstrated that predation by the common house gecko has a much more devastating effect on insect populations than does predation by the mourning gecko.

Brown anole (Anolis sagrei)

In North America, introduced *A. sagrei* preys on native lizards (Campbell 1999/2000, 2000; Gerber and Echternacht 2000; Vincent 2002). Since its introduction, *A. sagrei* has been expanding its range in North America and replacing native lizards *A. carolinensis* in Florida and *A. conspersus* in Grand Cayman Island as the common anole of urban environments and other open habitats (Gerber and Echternacht 2000). A review of intraguild predation (killing and eating among potential competitors) in *Anolis* lizards suggests that predatory interactions between anoles are relatively common, often asymmetric, and likely to affect the abundance and distribution of certain species (Gerber 1999). To assess the likelihood that predation of juvenile native anoles by *A. sagrei* adults is an important interaction in this process, Gerber and Echternacht (2000) assessed the propensities for intraguild predation and cannibalism for *A. sagrei* and *A. carolinensis* in Florida and for *A. sagrei* and *A. conspersus* in Grand Cayman. Predation experiments were conducted in cages, using freshly captured lizards, in which adult males of each species were presented with conspecific and heterospecific juveniles. Gerber and Echternacht (2000) found adult *A. sagrei* were significantly more likely to eat juveniles than were adult *A. carolinensis* or *A. conspersus*. The brown anoles were also significantly more likely to eat heterospecific than conspecific juveniles, whereas adult *A. carolinensis* and *A. conspersus* were not. Thus, the propensity for intraguild predation is asymmetrical in favour of introduced *A. sagrei* in Florida and Grand Cayman. The experimental cages artificially constrained juveniles, so it is not possible to extrapolate from these experiments to free-living populations. The authors recognised that further study is needed to determine the importance of intraguild predation by the brown anole under field conditions. Campbell (1999/2000) suggested that where dense shrub cover exists, *A. sagrei* should be able to co-exist with *A. carolinensis*.

Schoener and Spiller (1996) selected 12 subtropical small islands with web spider communities to study the impacts of *A. sagrei* introductions on resident spider communities. Four islands had natural lizard populations; the other eight islands did not. The islands with lizards had far lower spider densities and fewer spider species. Schoener and Spiller (1996) introduced three female and two male adult *Anolis* lizards on four islands and left four islands lizard-free. Within two years, the proportion of spider species becoming extinct on the four islands where lizards were introduced was 12.6 times higher than on the islands without lizards. Locally common and rare spider species were reduced by the introduction of lizards, but nearly all the rare spiders became permanently extinct. After two years, the density and number of spider species on the islands where lizards were introduced was no higher than on islands that had always had lizards. Schoener and Spiller (1996) concluded that predator introduction greatly threatens locally rare species and if these are regionally localised, threatens endangered species as well.

Spiller and Schoener (1998) conducted removal exclosure experiments with *A. sagrei* and found that the anole reduced the total number of individuals, species richness (number of species) and composite diversity of web spiders (prey species) compared to control exclosures with lizards present at natural densities. *A. sagrei* had the strongest influence on rare spider species. These results followed the same general pattern as an island introduction experiment, where introductions of *A. sagrei* resulted in rapid and permanent extinction of most rare web spider species, with only one web spider species ever persisting continuously on lizard-introduction islands (Schoener and Spiller 1996). After introduction of lizards to islands, mean density of web spiders (averaged over the last six years of the experiment) was five times higher on islands without lizards than on lizard-introduction islands. Spiller and Schoener (1998) suggested that in their mainland lizard exclosure experiments, spiders were being reintroduced from outside the exclosures but this happened less on isolated islands. There also may have been fewer refugia for spiders to escape lizard predation on island habitats.

Crested anole (Anolis cristatellus)

Fitch et al (1989) conducted a detailed field study of *A. cristatellus* (native to Puerto Rico) introduced to the Dominican Republic and found the introduced species displaces three native anoles: *A. distichus, A. chlorocyanus* and *A. cybotes*, by competition and/or by predation. According to Fitch et al (1989), the crested anole has become 'phenomenally abundant' in the Dominican Republic, but only occupies an area of about 160 square kilometres in an urban area and surrounding disturbed parks and gardens.

Other reptile species

Lever (2003) suggests introduced red-eared sliders (*Trachemys scripta*) could be a serious predator where there are rare amphibians in freshwater habitats.

Introduction of the curious skink (*Carlia ailanpalai*) to the Mariana Islands (specifically Guam) coincided with decline in populations of the Pacific blue-tailed skink (*Emoia caeruleocauda*) and possible eradication of Marianas blue-tailed skinks (*Emoia atrocostata*) and mottled snake-eyed skinks (*Cryptoblepharus poecilopleurus*) in the following decades (Rodda et al 1991).

According to Nevo et al (1972), the Italian wall lizard (*Podarcis sicula*) has spread and replaced the native Dalmatian wall lizard (*P. melisellensis*) throughout coastal areas and numerous islands in the Mediterranean.

According to Martínez-Morales and Cuarón (1999) the *Boa constrictor* snake was introduced onto Cozumel Island, Mexico, and is now widespread, and poses a threat to the existence of endemic terrestrial vertebrates on the island. According to Lever (2003), an anecdotal historical record suggested that the introduced *B. constrictor* on Cozumel Island 'severely affected' endemic fauna through predation, especially small animals living in the understorey. These species are now in 'very low' numbers, but there are no records of their abundance prior to the boa's introduction.

Predation by the introduced viperine snake (*Natrix maura*) in the Mediterranean Balearic Islands is believed to be a major cause of the decline of the native Mallorca midwife toad (*Alytes muletensis*, Alcover and Mayol 1981, Tonge 1986, Moore et al 2004ab).

Fritts (1993) suggested that the fauna of Christmas Island, which includes endemic species of reptiles, birds and mammals, could be threatened by predation by the lizard-eating wolf snake (*Lycodon aulicus*). Lever (2003) suggested the introduction of *L. aulicus* to the Mascarene Islands (Indian Ocean) probably contributed to the subsequent disappearance of half the island's lizards, including Bojer's skink (*Gonygylomorphus bogerii*).

Kraus and Cravalho (2001) suggested that several exotic snake species found in the wild in Hawaii could establish exotic populations and become significant predators of native forest and water birds, and *Thamnophis* snakes could also prey on native stream-dwelling fish such as gobies. These exotic snakes include *Boiga irregularis, Boa constrictor, Coluber constrictor, Python regius, Python molurus, Elaph guttata, Thamnophis* spp, *Lampropeltis* spp and *Pituophis* spp. Kraus and Cravalho (2001) further suggested that the dense populations of exotic prey species on Hawaii would make it easy for these snakes to establish and to maintain high population densities that would increase the risk to native prey species. Several authors also suggest there are hundreds of other snake species worldwide that could have similar devastating effects to *B. irregularis* on the naive native faunas of oceanic islands (Rodda et al 1997, Kraus and Cravalho 2001, Loope et al 2001).

Risk assessment significance: Predation by exotic reptiles and amphibians leads to reduced survival rates of prey species and has the potential to be highly detrimental to native species.

3.8.3 Habitat and ecological community impacts

(i) Community impacts of predation

Predation in aquatic communities is widely considered to be of profound importance in structuring prey species diversity, species composition, distribution, feeding and activity levels and production rates (Measey 1998bc). According to Measey (1998c), predation by aquatic predatory amphibians such as African clawed frog *X. laevis* has the potential to have major impacts on freshwater ecology, particularly when this species is present in high densities. *X. laevis* is a generalist predator that consumes a wide variety and size of invertebrate prey. Lobos and Measey (2002) found exotic *X. laevis* in Chile at high densities in some locations and suggested that at these densities, predation will have a significant impact on prey populations, and possibly result in trophic cascade effects, altering native species diversity and composition. There could also be secondary impacts resulting from increased water turbidity and nutrient release due to *X. laevis* disturbing sediments, and from a change in population dynamics of native predators.

Kraus et al (1999) speculated on the potential impacts of the *Eleutherodactylus* (*E. coqui* and *E. planirostris*) frogs in Hawaii and suggested their presence could reduce the abundance of native arthropods leading to increased pressure on the native avifauna, which depends solely on a diet of native insects. Tummons (2003) suggested dense populations of coqui frogs may eat over 200 kilograms of arthropods per hectare per year.

Dial and Roughgarden (1995) found experimental exclusion of *Anolis* lizards from rainforest canopy significantly increased arthropod abundance, which in turn significantly increased the level of herbivore damage on new leaves.

Spiller and Schoener (1997) compared damage to leaves of sea grape (*Coccoloba uvifera*) on seven islands without diurnal lizards *A. sagrei* and 11 islands with lizards. Damage was significantly higher on islands without lizards. These lizards are insectivorous and eat the insects that eat the leaves. Schoener and Spiller (1999) selected 12 islands (40–179 square metres) with shrubby vegetation within a 3.2 by 2 kilometre area in the Bahamas. Four islands had *A. sagrei* naturally present. Four of the eight islands without lizards were randomly selected and *A. sagrei* was introduced to these four islands. Over seven years, the effects of lizards on shrub herbivory and arthropods were monitored. Lizards indirectly reduced leaf damage and increased the number of small aerial arthropods towards the end of the seven-year study. Lizard introduction directly and rapidly reduced spiders to a similar density found on the natural lizard-containing islands.

In the Bahamas, introduced *A. sagrei* populations devastated spider and insect populations and had major top-down effects on food webs (Schoener and Spiller 1996, 1999; Spiller and Schoener 1997, 1998; Campbell and Echternacht 2003).

(ii) Provide prey for exotic predators

Kraus et al (1999) suggested exotic *Eleutherodactylus* frogs (*E. coqui* and *E. planirostris*) in Hawaii could provide an abundant food source for introduced predators, such as rats, cats and mongooses (*Herpestes javanicus*), leading to an increase in predator abundance and hence increasing the threat they pose to native forest birds. Kraus et al (1999) also suggested the frogs could provide an abundant food source for more damaging potential invaders, such as the brown treesnake (*B. irregularis*), if they are introduced to Hawaii. Similarly, Zug et al (1975) suggested that cane toads (*B. marinus*) may indirectly impact on the Hawaiian ecosystem by creating another food source for invasive predators such as mongooses and rats.

According to Fritts and Rodda (1998), introduced *Anolis carolinensis* lizards in Guam provide prey for the introduced brown treesnake (*B. irregularis*) and hence potentially enabled this snake to reach higher densities, which may have had flow-on ecosystem consequences. Similarly, Campbell (1996) suggested the introduction of the curious skink (*Carlia fusca*) to the Mariana Islands (Guam) may help maintain high densities of introduced *B. irregularis* by providing prey for it, with a consequent increased threat to native birds (Fritts and Rodda 1998, Rodda et al 1999).

(iii) Provide prey for native predators

Exotic reptiles and amphibians may also provide prey for native predators and this may increase the abundance of the native species. For example, Wilson and Porras (1983) suggest that populations of *Elaphe guttata* snakes have increased in some urban areas of Florida because increasing numbers of exotic lizards (*A. sagrei*) are available as prey.

In North America, the introduced brown anole (*A. sagrei*) is both predator and prey for native species and also competes with them (Campbell and Echternacht 2003). *A. sagrei* hatchlings are consumed by native anoles, which could lead to bottom-up effects on food webs (Campbell 2000, Gerber and Echternacht 2000). Birds are well known predators and competitors of anoles (Adolph and Roughgarden 1983, Waide and Reagan 1983) and the native black racer snake (*Coluber constrictor*) is also predator of anoles (Campbell 2000). Brown anoles and green anoles (*A. carolinensis*) overlap extensively in their diets (mainly arthropods), and adult green anoles are known to consume brown anole hatchlings (Campbell 2000, Gerber and Echternacht 2000). It is therefore likely that brown anoles have both top-down (mainly on insects) and bottom-up effects on food webs in areas where they are introduced.

(iv) Habitat alterations

Lobos and Measey (2002) suggest high densities of introduced *X. laevis* frogs in Chile disturb sediments and increase water turbidity, and this could have secondary impacts on other biota.

Searle (1980) reported that introduced bullfrog tadpoles (*Rana catesbeiana)* significantly reduced rates of phytoplankton primary production, altered species composition, and shifted the state of nitrogen in a pond from particulate to dissolved. Kupferberg (1997) found *R. catesbeiana* tadpoles introduced in a northern California river system significantly affected benthic algae, although effects varied across sites.

In Guam, Perry and Morton (1999) found regeneration rates of the woody vegetation following major disturbance was slow in areas where the seed bank had been removed. They said this was consistent with an absence of vertebrate seed dispersers due to predation by the treesnake *B. irregularis*.

(v) Indirectly facilitate survival of other exotic species

Simberloff and Von Holle (1999) and Richardson et al (2000b) describe positive interactions among non-native species that can exacerbate the problem of invasions, but these interactions have been poorly studied. Adams et al (2003) found that invasion of bullfrogs is facilitated by the presence of co-evolved non-native fish, which increase tadpole survival by reducing densities of predatory macroinvertebrates. Native dragonfly nymphs completely eradicated bullfrog tadpoles in a replicated field experiment in Oregon, United States, unless a non-native sunfish (*Lepomis macrochirus*) was present to reduce the dragonfly density. This pattern was also evident in pond surveys, where the best predictors of bullfrog abundance were the presence of non-native fish and water depth (Adams et al 2003). This study is the first experimental evidence of facilitation between two non-native vertebrates and supports the invasional meltdown hypothesis. Such positive interactions among non-native species have the potential to disrupt ecosystems by amplifying invasions, and Adams' et al (2003) study shows they can occur via indirect mechanisms.

(vi) Changes to community dynamics

Crossland (2000) studied the direct and indirect effects of the introduced cane toad (*B. marinus*) on populations of native frog larvae (*Limnodynastes ornatus* and *Litoria rubella*) in Australia. *B. marinus* eggs and hatchlings are highly toxic to predatory native tadpoles. Under 'naturalistic' conditions, populations of predatory *L. ornatus* tadpoles experienced significantly reduced survival when exposed to cane toad eggs and hatchlings. The toxic effects of *B. marinus* on *L. ornatus* indirectly facilitated the survival of later-breeding *L. rubella,* by altering predator–prey interactions between the two frog species. *L. ornatus* tadpoles are voracious predators of *L. rubella* eggs and hatchlings, so the reduction of *L. ornatus* tadpole populations by cane toads in turn reduced the intensity of predation by these tadpoles on *L. rubella* eggs and hatchlings, thereby increasing *L. rubella* survival. Crossland's (2000) results demonstrate that *B. marinus* may have both negative and positive effects on populations of native anuran larvae and thereby plays an important role in re-structuring native larval anuran communities via direct and indirect mechanisms.

Risk assessment significance*:* Changes to community dynamics, including secondary or flow-on effects in food webs, are the least studied and most difficult to predict effects of exotic reptiles and amphibians introductions. Exotic reptiles and amphibians also have the potential to have detrimental effects on recipient ecosystems when they alter the habitat of native species.

3.8.4 Potential to cause injuries

The following attributes give exotic reptiles and amphibians the potential to cause injury:

(i) Venomous or toxic bite

The bite of some snakes is venomous and the bite of some lizards can lead to blood poisoning.

For example, the brown treesnake (*B. irregularis*) caused considerable emotional trauma to residents and visitors alike when the snakes invaded human habitats with the potential for dangerous venomous bites to small children (Fritts et al 1990, 1994; Fritts and Leasman-Tanner 2001). However, there have been no human fatalities from brown treesnake bites (Fritts et al 1994).

(ii) Poisonous skin glands

In Florida, the introduced Cuban treefrog (*O. septentrionalis*) has toxic skin secretions that may irritate the mucous membranes of predators.

The cane toad (*B. marinus*) is well protected at all life stages by skin glands that secrete a highly toxic fluid. Animals that are not adapted to handle its toxicity can be killed when they attempt to eat the toad, its tadpoles or eggs (McCoid 1995, Crossland 2000). Domestic pets, mainly cats and dogs, have been killed by cane toad toxin (Freeland 1984, Lever 2003). Dogs are known to die within 15 minutes of mouthing a cane toad.

Much evidence of the impacts of the cane toad in Australia is anecdotal, with little data to support or refute the claims of negative impacts on native fauna at the population level (Freeland and Martin 1985). However, many native predators in Australia and elsewhere are susceptible to cane toad toxin. Varanids and other large lizards, some snakes and quolls appear to be particularly susceptible, and their populations may be threatened following cane toad invasion of an area (Freeland 1984). Australian native animals that have died from ingesting *B. marinus* include the Western quoll (*Dasyurus geoffroii)*, numerous snake species, crows, kookaburras, and the Tasmanian devil (*Sarcophilus harrisii,* Covacevich and Archer 1975). In some areas there have been drastic decreases in quoll and monitor populations following the cane toad's colonisation of their habitats (Clarke et al 2001). Burnett (1997) presented reliable anecdotal information that colonising cane toads in northern Queensland caused severe population declines in five predator species: the Northern quoll (*Dasyurus hallucatus*)*,* and the monitors *Varanus gouldii, V. mertensi, V. panoptes*, and *V. timorensis similis*.

Catling et al (1999) assessed the effects of expanding populations of *B. marinus* in the Northern Territory of Australia on the relative abundance and diversity of native fauna, before and after invasion by the toads. Four native vertebrate groups were sampled: amphibians (14 species), reptiles (46 species, of which 19 may eat cane toads), birds (171 species, of which 62 may eat cane toads) and mammals (17 species, of which eight may eat cane toads). In the short term, only the dingo (*Canis lupus dingo*) population was negatively affected.

Phillips et al (2003) predicted that predation by cane toads has the potential to have significant impacts on some Australian snakes and suggested that cane toads threaten populations of approximately 30% of terrestrial Australian snake species. Crossland (2000) found that under 'naturalistic' conditions, populations of predatory native tadpoles (*Limnodynastes ornatus*) experienced significantly reduced survival when exposed to cane toad eggs and hatchlings.

Cane toads are also thought to affect native amphibian populations and other aquatic fauna in Australia, mainly due to their toxicity (Freeland and Martin 1985, Crossland 1997, Crossland and Alford 1998, Crossland and Azevedo-Ramos 1999, Foulis and King 1999). Some native Australian fish (eg firetail gudgeon, *Hypseleotris galli*) avoid eating cane toad tadpoles or die if they do eat them (Freeland 1984).

Animals may learn or evolve traits to avoid cane toad poisoning, so the effects of the toxin may be temporary. For example, Crossland (2001) found that two species of predatory native Australian fishes (barramundi, *Lates calcarifer,* and sooty grunter, *Hephaestus fuliginosus*) learn to avoid toxic larvae of *B. marinus*. Individuals of both fish species recognised and avoided tadpoles one day after trial encounters, and no fish died in these trials. Phillips and Shine (2004) found two Australian snakes species (*Pseudechis porphyriacus* and *Dendrelaphis punctulatus*), whose range has been invaded by exotic cane toads, have evolved traits that make them less susceptible to cane toad poisoning: reduced gape size and increased body length. Gape size restricts the size of toad a snake can eat and thus the probability of eating a cane toad large enough to be fatal. These traits had evolved more strongly in snake populations that had been exposed to toads for longer periods. In Florida in the United States, *B. marinus* is prey for some birds, snakes and fish, but because there are two native *Bufo* in Florida, these predators have evolved methods to cope with *Bufo* toxins (Lever 2003). Domestic cats declined when *B. marinus* first arrived in Dumaguete City in the Philippines, due to cane toad poisoning, but they have since learnt to avoid cane toads and their numbers have recovered (Rabor 1952, Alcala 1957). Crossland and Azevedo-Ramos (1999) offered dead *B. marinus* tadpoles as food to native tadpoles species from Brazil and Australia. The native tadpoles from Brazil ate the dead *B. marinus* tadpoles without apparent ill effects, whereas the majority of the native Australian tadpoles died after eating them. Apparently, the tadpoles from Brazil, which had co-evolved with *B. marinus,* had developed resistance to cane toad toxins.

(iii) Organs and/or body size capable of causing physical injury

For example, crocodiles may be over six metres long, weigh up to 1000 kilograms, have strong jaws capable of crushing and have teeth capable of tearing flesh.

(iv)Traffic hazard

In Australia, *B. marinus* toads are considered to be a traffic hazard as their squashed bodies are slippery, causing vehicles to skid (Freeland 1984).

Risk assessment significance: Reptiles and amphibians that cause poisoning and/or physical injuries elsewhere in their range may be expected to have similar effects if they are introduced to Australia.

3.8.5 Role as disease carriers and reservoirs

Diseases spread by exotic reptiles and amphibians to native species may have ecological consequences (Daszak et al 1999, Garner et al 2006). Exotic reptiles and amphibians can serve as hosts, reservoirs and vectors for diseases and parasites that affect human and animal health. Three examples are listed below.

Red-eared slider (T. scripta)

The United States Food and Drug Administration has banned the sale of turtles under four inches in length because they can transmit the disease salmonellosis, which can be transferred to humans via drinking water (Newberry 1984, United States Geological Survey 2003d). Once enormously popular in the United States as pets, millions of red-eared sliders were sold domestically until this ban was applied. Millions are still exported each year to countries not so concerned about salmonella.

Introduced turtles also have the potential to introduce diseases to native fauna. Populations of *Actinemys marmorata* pond turtles in Washington were decimated by a respiratory infection in 1990, and introduced *T. scripta* were implicated as a likely vector for the infection (Hays et al 1999). Spinks et al (2003) found a female *T. scripta* in a California waterway showing signs of disease-related mortality and suggested the continual release of non-native turtles creates a high probability that diseases will also be introduced.

Cane toad (B. marinus)

According to Freeland (1984), cane toads in Australia eat human faeces and may thus spread parasites such as human-infesting worms (*Trichuris trichiura, Schistoma mansoni* and possibly human hookworms), canine *Uncinaria* hookworms and *Salmonella* bacteria. In American Samoa, it has been suggested that high densities of *B. marinus* may contribute to the high incidence of polluted drinking water and dysentery (Lever 2003). There has also been concern that *B. marinus* may carry parasites or diseases that can be transmitted to native fauna (Freeland et al 1986, Delvinqueir and Freeland 1988, Boland 2004). Large numbers of potentially pathogenic disease organisms have been isolated from cane toads (Speare 1990). On St Lucia in the West Indies, *B. marinus* are claimed to harbour ticks that affect cattle (Lever 2003). Whether diseases or parasites carried by *B. marinus* have negative effects on native species has not been investigated (Boland 2004).

African clawed frog (X. laevis)

Lobos and Jaksic (2005) suggest invading *X. laevis* in Chile could spread diseases to native anurans. *X. laevis* is now claimed to be the original source of the *Batrachochytrium* fungus that has been decimating native frog populations in many countries including Australia, North America and Central America (Weldon et al 2004).

Risk assessment significance: It is difficult to predict the role exotic species may have as vectors or reservoirs of diseases or parasites in new environments. However, species that harbour or transmit diseases or parasites elsewhere may transmit the same or similar diseases or parasites if these are present in Australia.

3.8.6 Hybridisation with native species and other genetic changes

When exotic reptiles and amphibians hybridise with native species and produce fertile offspring, this hybridisation corrupts the gene pool of the native species and hence may pose a threat to their survival (Arntzen and Thorpe 1999, Riley et al 2003, Storfer et al 2004).

Even a few exotic escapees can be sufficient to spread new, detrimental genes through native populations (Ebenhard 1988). A lack of reproductive isolation between exotic and native species can lead to genetic swamping, loss of native genetic diversity, and, in rare or endangered species, extirpation or extinction (Riley et al 2003). Rhymer and Simberloff (1996) suggested risks are highest when rare species hybridise with an abundant species, producing offspring that are fertile and can back-cross (introgress). Even without introgression, hybridisation may threaten the existence of rare species.

Hybrids may be produced spontaneously and survive in the wild. Such hybrids may be better adapted to survival and breeding than parent stock and may be more invasive (Lewontin and Birch 1966). Through the removal of geographic barriers that normally prevent mixing of taxa, or under pressures exerted through introductions that change normal behaviour patterns, hybrids can arise between species or genera that would not otherwise interbreed (Elvira 2001).

Butterfield et al (1997) considered hybridisation associated with released exotic reptiles and amphibians is a valid concern where these species are close relatives of native species. However, there are few proven examples, and only one well-documented study was found of an exotic amphibian hybridising with a native amphibian in the field and producing fertile progeny. Storfer et al (2004) and Riley et al (2003) examined hybridisation between a declining native salamander (the California tiger salamander, *Ambystoma californiense*) and an introduced congener (*A. tigrinum*). *A. californiense* is restricted to central California where *A. tigrinum* has been deliberately introduced as fish bait. Riley et al (2003) tested mitochondrial DNA and found hybrids present in six sampled ponds. These hybrids were viable and fertile. Despite a relatively ancient split and wide genetic divergence between these taxa, they were evidently interbreeding and threatening the genetic purity of the native species. Four artificial ponds showed greater genetic mixing than two natural ponds.

Other possible examples of hybridisation occurring in exotic reptiles and amphibians include:

- Capula (1993) and Capula et al (2002) found genetic evidence of past hybridisation between the introduced Italian wall lizard (*Podarcis sicula*) and the native wall lizard (*P. raffonei*) in the Aeolian Islands (Mediterranean), based on electrophoretic examinations. These authors suggested that *P. sicula* reduced the range and eradicated many populations of *P. raffonei* partly through hybridisation, but competition between the two species probably also played a significant role.
- Lever (2003) reported that introduced *Iguana iguana* on Guadeloupe (West Indies, South America) have almost replaced the native *I. delicatissima* iguana, partly through interbreeding that resulted in sterile hybrids and rapidly reduced numbers of the native species.
- Butterfield et al (1997) suggested that hybridisation may have occurred between native and introduced sub-species of *Anolis distichus* lizards in Florida.
- Lever (2003) reported that introduced red-eared sliders (*T. scripta*) are hybridising with introduced *T. decussata* turtles on Grand Cayman in the West Indies.
- Gorman and Atkins (1968) and Gorman et al (1971) suggested that introduced *Anolis aeneus* lizards in Trinidad are hybridising with introduced *A. trinitatis.*

Exotic species can have genetic effects other than hybridisation. They may have indirect effects by altering native species' patterns of natural selection or gene flow, in communities where they are introduced (Parker et al 1999). Competition, predation, or habitat alteration caused by exotic species may lead to changes in native species populations, including reduced population size, or reduced numbers of subpopulations or phenotypes, and this in turn can lead to changes in the genetic structure of the affected native species populations (Elvira 2001).

Risk assessment significance: Exotic species that have close relatives among Australia's endemic reptiles and amphibians could hybridise with these native species and corrupt their gene pool.

3.8.7 Social and economic impacts

(i) Species with known economic impacts

For most invasions of exotic reptiles and amphibians, there are little or no economic data available. The following species have demonstrated economic costs:

Coqui (E. coqui)

This tiny frog from Puerto Rico has loud, piercing calls that can measure 90–100 decibels at a distance of 0.5 metres. It exists in high densities where it has been introduced in Hawaii, and its calls are a problem for local residents and hotel guests, who complain about the noise keeping them awake at night (Kraus et al 1999, Kraus and Campbell 2002, Kaiser and Burnett 2006). Residents are encountering reduced property values and increased difficulty selling property (Kraus and Campbell 2002, Kaiser and Burnett 2006). This problem also occurs in other areas where *Eleutherodactylus* species have been introduced: for example, in French Guiana in South America, the calls of introduced *E. johnstonei* are disturbing the sleep of local residents (Lever 2003).

According to Kraus and Campbell (2002), the presence of the frogs in Hawaii may lead to rejection by trading partners of goods that may be infested with the frogs or their eggs. For example, Guam requires treatment of nursery products coming from Hawaii because of possible receipt of *E. coqui* in such shipments (E. Campbell, personal communication, United States Fish and Wildlife Service, Honolulu, 2005). Another negative consequence of the spread of *E. coqui* in Hawaii is the residents' illegal use of toxic chemicals in attempts to kill the frogs (Kraus and Campbell 2002).

Brown treesnake (B. irregularis)

Due to their arboreal nature, treesnakes climbing on electrical lines have become a huge economic burden to Guam. The snakes short out electrical systems and cause extensive electrical damage. This activity affects private, commercial, and military activities and causes damage totalling millions of dollars annually (Fritts and Leasman-Tanner 2001). *B. irregularis* also causes substantial losses to the poultry industry in Guam (Fritts and McCoid 1991).

Other snake species

Other exotic partially arboreal snakes, such as *B. constrictor, Python regius, P. molurus* and *Elaph guttata*, could cause similar economic damage to electrical industry infrastructure as that caused by *B. irregularis* on Guam, through short-circuiting powerlines (Kraus and Cravalho 2001). These authors also suggested that exotic snakes could also inflict substantial damage to the poultry industry in Hawaii if they establish.

Cane toad (B. marinus)

Cane toads have a number of documented economic impacts, particularly on water supplies. In Japan, introduced *B. marinus* pollutes freshwater with eggs and tadpoles (Lever 2003). Similarly, in Bermuda, where the sole source of fresh drinking water is rainwater tanks, introduced cane toads enter and drown in these tanks, thus polluting the water. In Australia, cane toads pollute and block water supply, drainage and storage facilities including swimming pools, their decomposing bodies pollute other water bodies, and they cause erosion of earth dams and creek banks by burrowing — it is costly to toad-proof these structures (Freeland 1984). It has also been suggested that animals killed by cane toad toxin may pollute drinking water supplies. For example, the Australian Northern Territory's Power and Water Corporation says cane toads threaten the quality of drinking water in Darwin: 'The impact is really on the native animals, particularly the small crocodiles that may eat cane toads and we don't want dead animals, as you would expect, being a threat in our catchments' (Day 2005).

In Australia and Bermuda *B. marinus* is a pest to apiarists because it preys on bees (Freeland 1984). Apiarists report that cane toads are often observed congregating around the entrances to hives where they take bees coming and going from the hive. In this situation one cane toad may consume as many as a hundred bees per day, leading to losses in production exceeding one million dollars per year (Freeland 1984, Clarke et al 2001).

In Barbados, cane toads are considered a pest in nurseries because they bury into potting mix and destroy seedlings (Lever 2003). Cane toads also damage seed beds in Grenada (Lever 2003). On St Lucia in the West Indies, *B. marinus* are claimed to trample commercial lettuce beds (Lever 2003).

Cuban treefrog (O. septentrionalis)

In the West Indies, introduced *O. septentrionalis* cause economic impacts by invading drinking-water tanks, cisterns and toilet vent pipes (Lever 2003).

African clawed frog (X. laevis)

X. laevis is an economic pest in its native southern Africa, where it spreads through disturbed habitats and interferes with aquaculture (Lafferty and Page 1997).

(ii) Potential damage to aquaculture facilities

Although no reports of harm to aquaculture facilities caused by exotic amphibians were found in the literature, the potential for such harm does exist. For example, *R. catesbeiana* bullfrogs caused considerable economic damage to a fish hatchery in Missouri (Corse and Metter 1980). The fish hatchery consisted of 400 ponds located in stream valleys raising goldfish (*Carassius auratus*) for aquarium trade and golden shiners (*Notemigonus chrysoleucus*) for fish bait. Stomach analyses of the bullfrogs showed that these fish (both species) were the highest volume of food eaten. In the goldfish ponds, each bullfrog on average ate $US12 worth of goldfish each year. There were 10 adult frogs per hatchery pond and 350 goldfish ponds, bringing the total cost of goldfish losses to $US42 000 per year for this hatchery. Tadpoles in the hatchery ponds also ate the commercial food provided for the fish. Although *R. catesbeiana* is native to Missouri, this species has established exotic and translocated populations in many countries around the world, where it presumably could inflict similar damage.

Risk assessment significance: Introduced reptiles and amphibians may bring economic benefits or cause economic harm. Because the distribution and abundance of introduced reptiles and amphibians are hard to predict accurately, forecasting the economic consequences of reptile and amphibian introductions to Australia is difficult. An examination of the economic consequences of previous introduction of a species elsewhere in the world, and any economic harm they cause in their native range, may provide some indication of potential economic consequences if a given species is introduced to Australia.

3.8.8 Other factors

The following factors have been suggested in the literature as potentially influencing the probability of impacts caused by exotic reptiles and amphibians:

(i) History of being a pest overseas

Daehler and Gordon (1997) suggest that 'the strongest predictor of negative impacts of a non-indigenous organism remains whether it has had negative impacts in other areas to which it has been introduced'. Reptiles and amphibians that are pests overseas may well become pests if they establish in Australia. Simple predictions can be made by assuming that invaders will cause significant impacts in a new area where they have established if they have already done so in other regions (Townsend and Winterbourn 1992, Ricciardi and Rasmussen 1998).

While correlative analyses are often limited by a scarce amount of comparable quantitative data, they can give an indication of potential impacts (Ricciardi and Rasmussen 1998). However, a species' history of impacts elsewhere is not an infallible guide to its potential impact in Australia. There are many examples in the scientific literature of species that have developed new behaviour and new dietary preferences when introduced to new environments. Such species hence had impacts that could not have been predicted from their history. So, species that have few harmful effects in their native (or previously introduced) range may have devastating effects when introduced to a new country (Bomford 2003, Hayes and Sliwa 2003). A further problem is that many potential pest species have not been introduced outside their natural range, and so have not had the opportunity to demonstrate their pest potential.

Risk assessment significance: Descriptive information on the impacts of previous invasions may provide a basis for useful predictions, although with a high degree of uncertainty. A precautionary approach is advisable for reptiles and amphibians species that have no history of establishing outside their natural range.

(ii) Rate of spread

Species that spread rapidly from their initial place of establishment are likely to be harder to eradicate, contain or control, and may be more likely to become widespread and be considered pests, than species with a slow rate of spread. The factors that influence the rate of spread and the final geographic range of an exotic species established in a new environment may differ from the factors that influence the probability of the initial establishment (Duncan et al 2001, Kolar and Lodge 2002, Forsyth et al 2004).

Risk assessment significance: There are inadequate data on rates of spread to enable this factor to be used to confidently predict the pest potential of future reptile and amphibian introductions to Australia. However, reptiles and amphibians that are known to have spread rapidly following their release into new environments overseas should be considered to pose a high risk because this trait is likely to make their eradication or control more difficult.

(iii) Taxa

Insufficient data are available to determine which exotic vertebrate reptile and amphibian families pose a high level of risk to native species and the environment, based on their history of impacts elsewhere.

There are, however, some species with a record of having significant detrimental impacts on native species, including extinctions, where they are introduced. For example, Lowe et al

(2004) published a list of the world's worst 100 invasive alien species and this list includes three amphibian species (*R. catesbeiana* bullfrog, *B. marinus* toad and *E. coqui* tree frog) and two reptile species (*B. irregularis* treesnake and *T. scripta* turtle).

Taxa that have novel adaptations (a possible example could be camouflage) that are not present in the native fauna of the region where they are introduced may prove problematic. For example, it is likely that many toad species, if introduced into Australia, could have negative effects, and the same would apply to introductions of chameleons throughout much of the world, or of frogs onto oceanic islands.

Taxa that reach high population densities (and hence high biomass densities) in their native or introduced ranges are more likely to have negative impacts. For example, all lizards in the genus *Anolis* and frogs in genus *Eleutherodactylus,* and many geckos (Family Gekkonidae), would fit into this category.

Risk assessment significance: Too little information is available in the scientific literature on the environmental, economic and social impacts of exotic reptiles and amphibians to enable a risk ranking at a taxonomic level higher than species. However, some individual species have clearly demonstrated their ability to have negative impacts in their introduced range. While, a species' history of impacts elsewhere is not an infallible guide to its potential impact in Australia, these species should be considered to pose a very high risk of impacts here. Taxa that have novel adaptations not present in the native fauna of Australia, and taxa that reach high population densities in their overseas ranges, may pose a higher risk of having negative impacts.

(iv) Abundance

Reptiles and amphibians reach densities among the highest recorded for non-congregating terrestrial vertebrates (Rodda et al 2001). Many high density records are from islands and most are of small species (Rodda et al 2001).

High density records include:

- about 67 600 per hectare for a Caribbean gecko (*Sphaerodactylus macrolepis*) in leaf litter on Guana Island (Rodda et al 2001)
- over 20 000 per hectare for coqui (*E. coqui*) in its native forests in Puerto Rico (Stewart and Rand 1991, Beard et al 2003) and up to 89 000 per hectare in its introduced range in Hawaii (Woolbright et al 2006)
- about 30 000 per hectare for North American red-backed salamander (*Plethodon cinereus*, Campbell and Echternacht 2003)
- 23 600 per hectare for barred anole (*A. stratulus*, Reagan 1992)
- at least 12 000 per hectare for brown anole (*A. sagrei*) introduced in Florida (Campbell and Echternacht 2003)
- at least 3700 per hectare for African clawed frog (*X. laevis*) in its introduced range in Chile (Lobos and Measey 2002)
- 46.3 per hectare (12 000 snakes per square mile) for the brown treesnake (*B. irregularis*) introduced on Guam (Fritts 1988).

Risk assessment significance: Species capable of reaching very high densities, and hence high biomass densities, can have strong top-down and bottom-up trophic level impacts on the ecological dynamics of the communities where they are introduced. Hence, species that are known to reach high densities either in their native or introduced ranges should be considered to pose a high risk of impact to Australia. However, species that have not attained high densities in their overseas ranges could still do so in Australia, so an absence of high density populations overseas should not be taken to indicate an absence of risk.

3.9 Discussion of factors affecting pest status for introduced reptiles and amphibians

Unfortunately, relatively little research has been conducted on the impacts of exotic reptiles and amphibians. Except for obvious species extinctions or economic losses, few studies have examined the possible suite of community changes that an invasive species can have. There are too few data to demonstrate how introduced species affect native species and thus it is not possible to make rational decisions about which species are safe to import because they pose a low risk of harm. This lack of reliable knowledge makes the development of a quantitative model for assessing the risks of impact for new species of exotic reptiles and amphibians in Australia unfeasible.

Of the hundreds of exotic reptiles and amphibians introduced around the world, only six species (*Boiga irregularis*, *Rana catesbeiana*, *Bufo marinus*, *Anolis sagrei*, *Osteopilus septentrionalis*, and recently *Trachemys scripta*) have been subject to even a modest degree of ecological research, and only the first three could be said to have been well studied in parts of their introduced ranges. This lack of attention may be due to exotic reptiles and amphibians not often being viewed as economic or agricultural pests, or even as ecological threats. Their ecological impacts on native species and communities are not usually obvious to people not trained in ecology, especially in comparison to other species such as large predatory mammals.

The impacts of exotic reptiles and amphibians are most readily recognised when an abundant introduced species leads to major declines in native species; for example the brown treesnake (*B. irregularis*) on Guam (Kraus and Campbell 2002). Less obvious and less studied impacts include:

- competitive interactions that limit resource availability to native species
- changes to food web structures
- genetic alterations
- changes in abundance of lower-order taxa and lower trophic-level species.

Defining harmful species and identifying species that cause or can potentially cause ecological harm is inevitably a subjective process (Hayes and Sliwa 2003). Ecological harm is difficult to define and evaluate when it refers to species that are of no direct economic value or to impacts on community structures and ecosystem processes. It is notoriously difficult to value components of native biodiversity or the benefits freely provided by ecosystem services that may be degraded by invasive species (Shine et al 2000). Such impacts are time consuming and hence expensive to evaluate, are often hampered by a lack of pre-invasion data, and therefore are largely under-reported in the scientific literature (Hayes and Sliwa 2003). Hence, some exotic species are perceived as having little obvious impact. There is no universally agreed formula to measure the environmental harm caused by introduced species and hence opinions on the type, extent and significance of impacts vary and even conflict (Hayes and Sliwa 2003). Techniques to assess the costs and benefits of alien species are evolving, but much research remains to be done, and some level of uncertainty will always exist.

Kraus and Campbell (2002) suggested the difficulty of observing and measuring trophic disruptions has restricted the study of reptile and amphibian invasions. These authors also suggested that failure to believe that small invading reptiles or amphibians can have significant ecological impacts has contributed to the failure by governments to implement eradication programs in the early stages of invasion — the time when successful outcomes can be achieved.

Even with the limited information available, it is clear that reptiles and amphibians have the same range of impacts as that reported for other exotic vertebrates. These impacts include:

- competition and hybridisation with native species
- predation on native species
- disruption of ecosystem trophic dynamics
- negative economic and social impacts (Kraus and Campbell 2002, Bomford 2003, Bomford and Glover 2004).

A more detailed review of the impacts of exotic reptiles and amphibians will soon be published by Kraus (in press).

In summary, the review of factors associated with adverse impacts presented in this chapter indicates that an increased risk is associated with reptiles and amphibians that have the following attributes/factors (with the caveat that reptiles and amphibians with an absence of these factors cannot necessarily be taken to pose a low risk of harm):

- have adverse impacts elsewhere
- have close relatives with similar behavioural and ecological strategies that have had adverse impacts elsewhere
- are dietary generalists
- stir up sediments to increase turbidity in aquatic habitats
- occur in high densities in their native or introduced range
- have the potential to cause poisoning and/or physical injury
- harbour or transmit diseases or parasites that are present in Australia
- have close relatives among Australia's endemic reptiles and amphibians
- are known to have spread rapidly following their release into new environments.

This list could be used as a checklist to make a qualitative assessment of the threat of impacts posed by the establishment of new exotic reptile and amphibian species in Australia. Such an assessment would be particularly desirable if decisions are being made on whether to import species that score a Risk of Establishment of Moderate or higher in the quantitative models presented in Sections 3.4–3.6.

4. Freshwater fish

Exotic fish species are commonly introduced for scientific, ornamental or recreational purposes (Pascual et al 2002, Bomford 2003, Cambray 2003, Canonico et al 2005). For example, tilapia (Family Cichlidae), have been intentionally dispersed worldwide for the biological control of aquatic weeds and insects, as baitfish for certain capture fisheries, for aquaria and as a food fish (Canonico et al 2005). This chapter reviews factors that affect the establishment success of introduced freshwater fish. It presents two models for assessing the risk of establishment. The first of these was developed by Bomford and Glover (2004) and Bomford (2006) for freshwater and estuarine species, based on previous introductions to Australia. The second model presented here is adapted from a generalised linear mixed model developed by Bomford et al (unpublished data) from data for exotic freshwater fish introductions to ten countries. Instructions for the use of these models are provided. A review of factors that affect the pest status of fish is also presented, with implications for risk assessment processes.

4.1 Factors affecting the establishment success of exotic freshwater fish

4.1.1 Key factors affecting establishment success

Factors affecting establishment success have been investigated for exotic freshwater fish introduced to Australia (Bomford and Glover 2004) and to ten countries (including Australia) by Bomford et al (unpublished data). These analyses were based on a database of introduction records collated by Arthington et al (1999), updated for more recent introductions to Australia. Relative to failed species, successful species:

- were introduced more times (had higher propagule pressure)
- had higher average climate matches to the countries where they were introduced
- were more likely to have established exotic populations elsewhere
- were more likely to belong to a genus or family that had higher success rates elsewhere.

These key factors and others that may contribute to establishment success of fish are reviewed below.

(i) Number of release events – propagule pressure

The release of large numbers of animals at different times and places (high propagule pressure) enhances the chance of successful establishment (Moyle and Light 1996b, Townsend 1996, Arthington et al 1999, Grevstad 1999, Kolar and Lodge 2001, MacIsaac et al 2001, Mack and Lonsdale 2001, Ricciardi 2001, Marchetti et al 2004, Leprieur et al 2008). Small populations (or small propagules of released animals) are more susceptible than large populations to extinction from factors such as increased risk of predation, not finding a mate, reduced breeding success, poorer hunting success or increased inter-specific competition (Soule and Simberloff 1986, Williamson 1989, Arthington et al 1999, Dennis 2002). Demographic stochasticity, such as random fluctuations in the proportions of males and females, will play a major role in determining the survival of small populations, particularly for short-lived or monogamous species (May 1991, Lande 1993, Legendre et al 1999).

Environmental stochasticity, including chance events such as droughts and floods, are also likely to drive small populations to extinction (MacArthur and Wilson 1967, Simberloff 1989, Stacey and Taper 1992, Caughley 1994). Small populations may also lose genetic variability, which may reduce the probability of long-term survival (Soulé 1987, May 1991). Ehrlich (1989) suggested that the release of more individuals may increase success rates because larger invading groups will have a greater pool of genetic variability. This variability might reduce founder effects and enhance the chances of rapid adaptive radiation in the new environment. The minimum viable population size for successful invasion is not known for most species.

Repeated releases over an extended period will increase the chance of successful invasion, simply because the release 'experiment' is repeated many times, under different biotic and abiotic conditions, including different climates and seasons, condition of released animals and numbers of natural enemies present.

Risk assessment significance: The number of release events is a significant predictor of establishment success, and the total number of individuals released and the number of sites at which releases occur may also affect establishment success. These three variables, which collectively determine the level of propagule pressure, should be considered as key factors when managing the risk of exotic species establishing in Australia. The number of fish that escape or are released is likely to increase if more species are kept, in higher numbers, and in more locations. Hence, propagule pressure can be reduced by restricting which species are kept in Australia, the number of collections holding a species, the number of individuals held in each collection, and the security conditions for keeping species. Educating people about the risks of releasing exotic fish into waterways will also help reduce the risk of establishing new exotic populations. Any changes to policy or management for exotic species that (i) allow more species to be imported, or (ii) reduce restrictions on where exotic species can be held or the numbers held, are likely to increase the risk that more exotic fish species will establish wild populations in Australia.

(ii) Climate match

A frequently stated hypothesis in the biological invasion literature is that species should have a greater chance of establishing if they are introduced to an area with a climate that closely matches that of their original range (Moyle 1986, Brown 1989, Williamson 1996, Davis et al 1998, Arthington et al 1999). Climate match is a measure of the similarity between the sites of origin and release based on rainfall and temperature data. The environmental condition of water bodies in a region is broadly determined by climate.

Arthington et al (1999) considered that temperature is the most limiting environmental factor in Australia's freshwater that affects exotic finfish establishment. Water temperature is a major determinant of whether exotic fish establish breeding populations (Nico and Fuller 1999).

Risk assessment significance: The level of climate match should be considered as a key factor when assessing the risk that new exotic species could establish. However, climatic matching only sets the broad parameters for determining if an area is suitable for an exotic fish to establish. Many factors, such as unsuitable water chemistry or flow dynamics, the absence of suitable spawning habitats or food, or the presence of competitors, predators or diseases, could prevent an exotic fish from establishing in a climatically matched area, so climate matching would overestimate the area of suitable climate in Australia. On the other hand, these same biotic and non-climate related abiotic factors could prevent a species from spreading to surrounding areas with suitable climate from its native or current introduced range (Taylor et al 1984) — in such a case, climate matching could underestimate the area of suitable climate.

(iii) History of establishing exotic populations elsewhere

A proven history of establishing exotic populations may indicate that a species has attributes that increase the risk of it successfully establishing in other areas (Ricciardi and Rasmussen 1998, Arthington et al 1999, Kolar and Lodge 2002, Hayes and Sliwa 2003).

Kolar and Lodge (2002) found that fish species that successfully established in the Great Lakes of North America were more likely to have been introduced successfully elsewhere, than fish that failed to establish. Bomford et al (unpublished data) found that freshwater fish with a high establishment success elsewhere were more successful at establishing in the ten studied countries than fish with a lower success elsewhere.

Risk assessment significance: Because a history of establishing exotic populations elsewhere is a significant predictor of establishment success for exotic fish introduced to ten countries and to the Great Lakes of North America, this variable should be considered as a key factor when assessing the risk that exotic fish could establish in Australia. However, many species that are potential exotics have not been transported to and released in new environments, so they have not had the opportunity to demonstrate their establishment potential. Hence, caution should be applied when using a history of establishment elsewhere to predict a species' establishment potential in Australia, if the species being assessed has little or no history of previous introductions.

(iv) Taxonomic group

Some ecologists consider that fish species that are closely related to fish with a history of being invasive present a higher risk of establishing in Australia (Moyle 1986, Arthington et al 1999). Daehler and Strong (1993) suggested this risk may be enhanced if the closely related species has similar habits to the known invader.

Moyle (1986) looked for patterns of fish introductions to North America and found that the majority of species that have become established outside their natural range come from the families Salmonidae, Cyprinidae, Ictaluridae, Poeciliidae, Cichlidae, Centrarchidae and Percidae. Moyle (1986) also noted that most other finfish families have at least one species that has been a successful invader.

Bomford et al (unpublished data) found both genus and family were significantly correlated with establishment success for 1634 introduction events of 280 species of freshwater finfish species introduced around the world.

Risk assessment significance: For fish species with a history of introductions to new areas, or with relatives in the same genus or family with such a history, previous establishment success rates should be considered a key predictor of future establishment success. A precautionary approach may be advisable for fish that have little or no introduction history, or have no relatives with an introduction history.

4.1.2 Other factors potentially affecting establishment success

Bomford and Glover (2004) and Bomford et al (unpublished data) did not test the influence of species-level factors (such as diet, offspring per year, growth rate, body size, lifespan or adaptation to disturbed habitat) on establishment success. However, many such factors have already been comprehensively investigated for freshwater fish by Kolar and Lodge (2002), Marchetti et al (2004) and Ruisink (2005) (Table 4.1). None of the ten species' attributes listed in Table 4.1 was found to be significantly associated with establishment success across more than one of these studies. Hayes and Barry (2008) found no species-level factors to be consistently associated with establishment success in other vertebrates or invertebrates.

Table 4.1 Species-level characteristics and establishment success for three studies comparing successful and failed introductions of exotic freshwater fish

Species' attribute	Influence of attribute on establishment success		
	Kolar and Lodge (2002)	**Marchetti et al (2004)**	**Ruisink (2005)**
Diet	No association	No association	Omnivores more successful than other diet groups
Adult body length	Shorter adults possibly more successful (P = 0.06)	No association	Shorter adults more successful
Longevity	Short-lived fish more successful	Long-lived fish more successful	-
Ecological tolerances	Fish with broad ecological tolerances marginally more successful (P = 0.07 for salinity; P = 0.10 for temperature)	Fish with broad ecological tolerances more successful	-
Parental care	No association	Fish with parental care more successful	-
Fecundity	No association	No association	-
Growth rate	Fish with rapid growth more successful	-	-
Year of introduction	No association	-	No association
Family	No association	-	Association
Reason for release	No association	Fish intentionally released to establish a wild population more successful	-

Dash = not tested.

There are many additional factors that are hypothesised to enhance the probability of establishment but for which scientific supporting evidence is currently lacking or equivocal. Eighteen such factors are listed below, with a brief assessment of their predictive value for risk assessment.

(i) Overseas geographic range size

Species that are widespread and abundant in their original range may be more likely to establish exotic populations than species with more restricted ranges (Ricciardi and Rasmussen 1998). Williamson (1996) suggested that a wide geographic range could indicate flexible or generalist species, or good dispersers, and hence species that are more likely to invade successfully. Exotic species with an ability to tolerate wide habitat and climatic variability may be more successful at establishing (Swincer 1986, Ehrlich 1989).

Bomford and Glover (2003) compared 31 successfully established species of exotic fish in Australia with 19 species that were introduced but failed to establish. They counted the number of grid squares (one degree latitude by one degree longitude) on a world map (excluding Australia) where Fishbase (2004) had occurrence records for each species. They found that the successful species were present in more than twice as many grid squares as the failed species.

Risk assessment significance: Overseas geographic range size could be considered a factor when assessing the risk that new exotic species could establish in Australia, although more work is needed to establish the significance of this factor for fish.

(ii) Rate of population increase (*r*) and related variables

Many ecologists consider that high fecundity and associated attributes (rapid growth rates and early sexual maturity, large clutch size, frequent spawnings and extended spawning period, high breeding frequency, short gestation and opportunistic or aseasonal breeding) contributes to successful vertebrate invasions (Taylor et al 1984, Moyle 1986, Kailola 1989, Fryer 1991, Crowl et al 1992, Lodge 1993a, Ricciardi and Rasmussen 1998, Arthington et al 1999, Elvira 2001, MacIsaac et al 2001). Such traits are often referred to as *r*-selected. The intrinsic rate of increase (*r*) of a species might be expected to determine the speed with which a small founding population can rise above the critical threshold number needed for demographic viability. Some ecologists suggest vertebrates with short generation times should be more successful invaders than those with long generation times (Ehrlich 1989, Lockwood 1999). In contrast, Crawley (1986) suggested high adult longevity ensures that offspring are produced over a protracted period, thus enhancing the probability of establishment by increasing the chances that offspring will encounter suitable conditions for establishment. Lodge (1993a) suggested *r* may not be an important determinant of invasion success for fish.

Kolar and Lodge (2002) found that fish that successfully established in the Great Lakes of North America had faster relative body growth rates (an *r*-selected trait) than fish that were introduced but failed to establish. Bruton (1986), however, found a more or less equal representation of *r*-selected and *K*-selected (slow growing, with a low rate of increase, long generation times) species in invasive fish species of South Africa.

Risk assessment significance: The evidence supporting a link between factors associated with a high *r* value and high establishment success is limited and equivocal. Data for measuring *r* are also unavailable for many fish species. Therefore, it is unlikely that factors associated with *r* will be useful for predicting the probability of establishment success at present.

(iii) Suitable site — resources enemies and 'biotic resistance'

The availability of habitat near the release site that meets a species' physiological and ecological needs is necessary for establishment (Welcomme 1988, Ross 1991). Introduced fish need refuges near the release site where they can obtain food, water, shelter and protection from natural enemies. Both habitat disturbance and an absence or low occurrence of natural enemies (predators, competitors, parasites or diseases) are often suggested to favour establishment (Mandrak 1989, Crowl et al 1992, Moyle and Light 1996a, Leprieur et al 2008). Fish that are ecologically or behaviourally distinct from fish in the recipient habitat may have an advantage in establishing either because the resident fish do not compete with them or are the losers in such interactions.

Some ecologists consider that biotic conditions in the recipient habitat play a major role in determining introduction success. In particular, natural enemies may resist invaders, so communities that are rich in diverse species may be more resistant (or exhibit 'biotic resistance') to invasion than species-poor communities (Welcomme 1988, Ross 1991, Lever 1996, Moyle and Light 1996a, Elvira 2001, Kennedy et al 2002, Fridley et al 2007). For example, based on his examination of 1354 introductions of 237 exotic fish species into 140 countries, Welcomme (1988) suggested that habitats with low levels of species diversity are more likely to be successfully invaded than more species-rich communities. He gives examples of two freshwater species-poor habitats where the establishment rate of introduced fish has been high. In freshwater systems on tropical islands, the introduction success rate of all fish was 73.3%, and in high-altitude lakes and rivers in the tropics the introduction success rate of salmonids was 73.1%. Moyle (1986) found similar high introduction success rates in species-poor communities in the western drainage systems of the United States. Similarly, Ross (1991) assessed 31 papers studying the introduction of exotic fish to 26 aquatic systems and found the establishment of exotic fish was higher in areas that had fewer native fish.

The importance of competition and disease as a cause of failure may be underestimated, because these factors are difficult to measure and so their effects are rarely assessed. Habitats where there are no resident species that have an ecological strategy similar to the introduced exotic species may be more likely to be invaded. This is because the new species may fill a 'vacant niche' without competition from species with similar ecological strategies.

Abiotic factors are major determinants of establishment success in aquatic systems. Ricciardi and MacIsaac (2000) pointed out that although examples exist where natural enemies have repelled invaders (ie where biotic resistance is high), many complex aquatic systems (which should also have high biotic resistance) have been invaded multiple times, such as Lake Victoria and the Caspian Sea. Similarly, Fryer (1991) cited examples of invasion by fish of Lake Malawi, which has a rich fish fauna and some of the most complex of all freshwater fish communities. This study showed how easily some highly diverse tropical ecosystems can be invaded. Moyle and Light (1996a) noted that exotic fish have become established in many lakes and streams that originally had no fish, as well as in complex assemblages with high species diversity. This observation conflicts with one of the most well established generalisations of the aquatic invasion literature, that systems with low diversity and complexity are the most susceptible to invasion (Lodge 1993ab). So, Moyle and Light (1996b) concluded that this frequently cited generalisation (Lodge 1993ab) is not supported by examples from aquatic systems. Based on their studies of invading fish in Californian streams and estuaries, Moyle and Light (1996ab) contended that all aquatic systems are invasible regardless of the biota already present, if abiotic conditions are appropriate. Ricciardi (2001) drew the same conclusion from studies of the invasion history of the American Great Lakes. Thus, it appears likely that the abiotic components of the environment have the principal role in determining establishment success.

The theory of biotic resistance may not be valid, at least for aquatic systems. The theory predicts that communities become more resistant to invasion as they accumulate more species, because species accumulate that have been successful competitors or predators, as demonstrated by the success of their original invasions (Case 1991, Moyle and Light 1996a). In contrast to this theory, some ecologists have suggested that invasions may be assisted by previous invasions, and that pre-established exotic species appear to facilitate the establishment of later-arriving exotic species (Simberloff and von Holle 1999, Ricciardi 2001). For example, Moyle and Light (1996b) suggested the invasion of the American Great Lakes by salmonids (*Oncorhynchus* spp) was greatly facilitated by disruption of the lake ecosystem by two previous invading fish, the sea lamprey (*Petromyzon marinus*) and the alewife (*Alosa pseudoharengeus*). Chronic exposure to introduced species thus subjects a community to 'invasional meltdown' (an accelerated rate of invasion), particularly when there are facilitative interactions between co-evolved invaders. Hence, the widely cited view in relation to terrestrial communities, that species-rich communities are resistant to invasion or become increasingly resistant with each species addition, is apparently invalid for aquatic systems subject to frequent human-mediated introductions (Ricciardi and MacIsaac 2000, Ricciardi 2001).

Because of these currently conflicting views of biotic resistance, it is not possible to draw general conclusions about the susceptibility of ecological systems to invasion based on their biotic components (Moyle and Light 1996b, MacIsaac et al 2001). Fridley et al (2007) reviewed the evidence supporting both positive and negative relationships between native biodiversity and the invasions of exotic species. These authors concluded that ecosystems rich in native species are also likely to be hotspots for exotic species, but that reduction of local species richness can further accelerate the invasion of these habitats.

The role of natural enemies in establishment success is difficult to measure, and limited quantitative evidence could be found to support this theory. Evidence is equivocal whether the level of species diversity or the presence of previous invaders in recipient habitat is correlated with introduction success.

Risk assessment significance: No consistent patterns between community structure and susceptibility to invasion have been demonstrated for fish. The potential relationships between an organism and possible parasites, predators, diseases and competitors are usually impossible to predict, except in a generalised, qualitative sense. These factors are difficult or expensive to measure quantitatively, so there is little evidence to support or reject their role in establishment success. Hence, these factors are unlikely to be of value for risk assessment and management. It would also be extremely difficult to rank habitat suitability objectively, so this factor probably also has limited value for quantitative risk assessment. An exception would be for separating disturbed habitat from undisturbed habitat and for climate matching. The significance of the availability of suitable microhabitats and microclimates for fish is largely unknown. Hence, it is difficult to quantify microclimate variables in a way that would be useful for managing the risk of species establishment.

(iv) Environmental tolerances for abiotic conditions

Fish species that are able to survive and reproduce under a wide range of conditions may be more likely to establish than less tolerant species (Taylor et al 1984, Arthington and Mitchell 1986, Kailola 1989, Pimm 1989, Crowl et al 1992, Ricciardi and Rasmussen 1998, Arthington et al 1999, Nico and Fuller 1999, Elvira 2001). In addition to climate-related factors such as temperature, environmental variables include hydrologic regime (water levels, flow, turbidity etc) water chemistry (oxygen levels, salinity, hardness, acidity, pollution etc) and substrate type (rocks, sand, mud, weed beds etc) (Moyle and Light 1996a, Arthington et al 1999). Cichlids, cyprinids and some poeciliids can survive for some time in water temperatures as low as 5^{o}C and as high as 43^{o}C, in freshwater and hypersaline waters, and in polluted and/or de-oxygenated waters (Taylor et al 1984, Arthington et al 1999). Elvira (2001) gives the examples of mosquitofish (*Gambusia affinis* and *G. holbrooki*) that can survive temperature ranges of 6–35^{o}C, extremely low oxygen concentrations and salinities twice that of seawater.

Harsh environmental conditions in relation to the physiological capabilities of a fish can override a good climate match, and make a particular water body unsuitable for a potential invader. Such conditions include high, low, extremely variable or unpredictable values of salinity, water hardness, turbidity and acidity.

Moyle and Light (1996ab) suggested that a close match between an invader's physiological and life history requirements and the abiotic components of the invaded system will determine invasion outcomes, regardless of biotic resistance. Climate matching is one component of the abiotic environment. Another abiotic component of an invaded aquatic system, which Moyle and Light (1996ab) considered plays a large role in determining invasion success, is the hydrologic regime. The hydrologic regime includes such factors as flow speeds, turbulence, depth, volumes and seasonal patterns, or unpredictable random changes to these factors. These may be broadly correlated with climate, but are also affected by factors such as altitude, geology, salt-water encroachment, catchment land use and human constructions (dams, channels, stream diversions and irrigation). Stream, lake bottom and bank structures and sediments may also be important. For example, trout have been unable to establish in many slow-moving Australian rivers within their natural thermal tolerance range because of the paucity of suitable spawning substrates (Weatherley and Lake 1967). With European carp, torrential waters and coarse substrates of river headwaters may act as barriers to upstream movement (Arthington et al 1999).

Species able to tolerate the high water temperatures and severe hypoxic conditions of many floodplain water bodies may be better invaders (Fryer 1991, McNeil and Closs 1998, Arthington et al 1999, Nico and Fuller 1999, Kailola 2000). Only anecdotal evidence was found to support a link between tolerance to hypoxic conditions and establishment success. For example, European carp and goldfish have high tolerance to hypoxia, the mosquitofish (G. holbrooki) has a particularly efficient use of aquatic surface respiration and the weatherloach tolerates hypoxia by gulping air at low oxygen levels (McNeil and Closs 1998) — these species are the most widespread of Australia's freshwater exotic fish (Kailola 2000). The ability to breathe atmospheric air is a huge advantage to dispersal, and its advantage is well illustrated by taxa such as walking catfish (*Clarias batrachus*), snakeheads (*Channa* spp) and climbing perch (*Anabas testudineus*) that have established in New Guinea (Kailola 2000). However, Fryer (1991) pointed out in contrast that the ancient lung fishes (Dipnoi) are also air breathers, but this taxon is not a good invader.

Kolar and Lodge (2002) found that fish species that successfully established in the Great Lakes of North America tolerated wider ranges of temperature and salinity than fish species that failed to establish.

A comparison of the tolerances for salinity, acidity and water hardness for exotic fish that established or failed to establish in Australia following their introduction indicated wide variability within both groups (Bomford and Glover 2003). There was no indication that the species that established had broader tolerances (Bomford and Glover 2003).

Moyle and Light (1996b) found that in Californian streams, invading fish are most likely to be successful if they are adapted to the local, highly seasonal stream flow conditions. Lodge (1993a) also found evidence supporting the importance of hydrologic regimes in the invasion success of fish in mid-western lakes in the United States.

Risk assessment significance: While climate matching can provide a broad envelope of suitable environmental conditions, fish also require suitable hydrologic regimes that meet their physiological and life history requirements. Only detailed studies of conditions where releases are going to occur will determine if such requirements are met. If such studies are not available, it is probably reasonable to assume that these requirements will be met, especially for fish species with broad environmental tolerances.

(v) Genotypic and phenotypic variability and behavioural flexibility

Animals with high genotypic and phenotypic variability may be more successful at establishing (Townsend 1996, Vermeij 1996, Ricciardi and Rasmussen 1998, Arthington et al 1999, Kailola 2000, Elvira 2001, MacIsaac et al 2001). Behavioural flexibility may also be an advantage. One of the chief reasons for the global success of brown trout is its polytypic nature — it naturally occurs as a series of reproductively isolated stocks each with slightly different characteristics (Townsend 1996). This is probably also the case for European carp and goldfish strains (Kailola 2000).

High genotypic and phenotypic variability in diet, behaviour and nesting habits in different environments may increase establishment success, because high variability increases the potential for rapid adaptive radiation. Fish are generally more plastic in their potential for hybridising than are mammals, and fewer crosses between fish species result in sterile progeny (Welcomme 1988). Hybrids may be produced spontaneously and survive in the wild. Such hybrids may be better adapted to survival and breeding than parent stock and be more invasive.

According to Arthington et al (1999), various studies have shown that some fish groups (notably cichlids and cyprinids) acclimatise over generations to 'less suitable' environments. However, no quantitative evidence could be found to support the theory that high genotypic and phenotypic variability enhances establishment success. Some successful animal invaders have in fact very low heterozygosity (Moller et al 1993).

Risk assessment significance: Fryer (1991) and Williamson (1996) considered that genetics will have little to offer for predicting the likelihood of establishment for exotic species. Fryer (1991) considered that genetic changes that take place in newly established populations reflect reaction and adaptation to the new environment rather than any genetic features favouring invasion.

(vi) Dispersal ability

Fish with good dispersal abilities may be better invaders perhaps because they are better able to seek out habitats suitable for survival and reproduction (Moyle 1986, Kailola 1989, di Castri 1990, Ricciardi and Rasmussen 1998, Arthington et al 1999, MacIsaac et al 2001). Moyle (1986) considered the ability to disperse rapidly from the point of introduction to be one of the most important characteristics of a successful introduced fish species. He found that impressive dispersal records existed for many of the introduced fish species of North America.

No quantitative evidence could be found to support this theory. Fryer (1991) was unable to find any empirical studies which demonstrated a consistent relationship between dispersal ability and invasion success. He pointed out that some fish species, like African catfish (*Clarias gariepinus*) and Nile tilapia (*Oreochromis niloticus*), which have seemingly poor dispersal abilities, have been hugely successful as invaders. Kolar and Lodge (2002) assessed the dispersal rates of 16 exotic fish species successfully introduced to the American Great Lakes and ranked seven species as slow spreaders and nine species as fast spreaders.

Risk assessment significance: Dispersal ability is generally a difficult trait to quantify and if evidence from fish invaders of the American Great Lakes is applicable elsewhere, it is unlikely to be a useful factor for predicting establishment success.

(vii) Broad diet

Some ecologists suggest animals with broad and/or flexible diets (dietary generalists) may be more successful at establishing exotic populations than those with restricted diets (dietary specialists). This is because their flexibility would enable them to exploit a greater range of food types than dietary specialists, so reducing the chances of food being limiting (Taylor et al 1984; Arthington and Mitchell 1986; Kailola 1989, 2000; Crowl et al 1992; Ricciardi and Rasmussen 1998; Arthington et al 1999; Nico and Fuller 1999; MacIsaac et al 2001). For example, the successful invader Mozambique tilapia is normally a herbivore–detritivore but is known to switch to carnivory in some circumstances (Arthington and Bluhdorn 1994).

No evidence was found in the literature supporting this theory for fish introductions. However, many of the exotic fish species that have been introduced both in Australia and overseas are dietary generalists, so analyses comparing successful and failed introductions lack statistical power to discriminate on diet. Hence the hypothesis remains largely untested and the role of a generalist diet in enhancing establishment success remains an expert opinion supported by many ecologists rather than an established relationship.

Risk assessment significance: Because many ecologists consider having a generalist diet increases the probability of establishment success, and because nearly all exotic vertebrates established in Australia do have generalist diets, this variable could be considered as a possible contributory factor when assessing the risk that new exotic species could establish here.

(viii) Ability to live in human-disturbed habitats — human commensalism

Many ecologists consider that an ability to live in human-modified or other disturbed habitats (human commensalism) is a major factor contributing to establishment success (Moyle 1986, Ross 1991, Lever 1996, Moyle and Light 1996ab, Williamson 1996, Ricciardi and Rasmussen 1998, Arthington et al 1999, MacIsaac et al 2001).

The success of human commensals may be partly due to many exotic animals coming from, and taking up residence in, human-modified habitats, where the types of food and shelter they are adapted to are present, so there is little need for their ecological niche to change for successful establishment. Exotic species that are pre-adapted to the types of habitat, food, shelter, predators or diseases present in Australia may be more successful at establishing.

Moyle and Light (1996a) predicted that in aquatic systems with high levels of human disturbance, a much wider range of species can invade than in systems with low levels of human disturbance. They also predicted that these invaders are much more likely to succeed. They suggested that this is because human-disturbed systems, such as reservoirs, tend to resemble one another over broad geographic areas, so introduced species may be pre-adapted to these types of habitats.

Disturbed habitats may be more susceptible to invasion than undisturbed ones for three main reasons. First, new, unoccupied niches may be created in disturbed habitats. Secondly, activities associated with water management may protect newly introduced small populations from environmental hazards, such as drought, flooding, parasites, predators and competitors. Such protection would allow them to grow to a size where they are not threatened with extinction by chance environmental events. Thirdly, disturbed habitats are often able to support a high level of species diversity because environmental variation prevents any one species from dominating other species (Connell 1978, Moyle and Light 1996b). Environmental patchiness can facilitate the coexistence of introduced species with potential competitors and predators (Anderson and May 1981, Crowl et al 1992).

Moyle and Light (1996b) suggested that the most invasible systems are those with intermediate levels of human disturbance. Frequent and unpredictable fluctuations in environmental conditions make it difficult for any one species or group of species to dominate the system. This allows the co-existence of species that might otherwise eliminate one another in more predictable systems, or that would be eliminated by environmental conditions in more highly disturbed/ altered environments (Moyle and Light 1996b). Changes in competitive ability can be related to environmental changes, so that in constantly shifting natural environments, no clear winner emerges (although exceptions are tropical lakes, desert springs or reservoirs, which are relatively constant environments). Similarly, Moyle and Light (1996b) suggested that the existence of large populations of predators can presumably prevent invasions, but only if the environment stays constant enough to maintain the large populations — unusual floods, droughts or human disturbance can upset this biotic resistance. In general, Moyle and Light (1996b) concluded that biotic resistance in the form of predators, competitors and diseases are less important than environmental resistance (habitat/climate matching) except perhaps in the early stages of an invasion, when numbers of the invader are low.

Many successful invaders use dispersal mechanisms that involve human activities. Hence, human commensals may have greater *opportunity* for establishing rather than having an intrinsic ability to be better at establishing.

Generalist invaders capable of withstanding disturbed conditions associated with urban and industrial pollution, and low oxygen levels, may be more successful than species that require better environmental conditions (Moyle and Light 1996b). This observation was made from the fact that many fish releases occur in disturbed and polluted waters in or near human settlements (Moyle and Light 1996b). Habitat disturbance and the modification of waterways and flow regimes have provided habitats for which introduced species are often better adapted than native fish (Arthington et al 1999).

Ross (1991) examined 31 papers studying the introduction of exotic fish to 26 aquatic systems and found establishment success was generally higher in systems disturbed by human activities. However, despite the high numbers of exotic species found in many highly disturbed aquatic systems (Moyle and Light 1996ab), there are also many records of exotic fish establishing in relatively pristine habitats. For example, Arthington and Bluhdorn (1995) recorded Mozambique tilapia established in relatively undisturbed areas of Queensland.

Moyle (1986) and Arthington et al (1990) reviewed the role of habitat disturbance in the establishment of exotic fish species in North America and Australia. They found that, although fish have been introduced to many environments, success was highest where waters had been dammed, diverted or otherwise modified to create reservoirs or more constant flow regimes.

Risk assessment significance: Because many ecologists consider an ability to live in disturbed habitats increases the probability of establishment, and because most successfully established exotic vertebrates are human commensals, this variable could be considered as a possible contributory factor when assessing the risk that new exotic species could establish here. However, it is necessary to recognise that while environmental disturbance may enhance probability of success, it is also possible for exotic fish that live in disturbed environments to establish in undisturbed areas. Moyle and Light (1996b) conceded that their finding that the most invasible systems are those with intermediate levels of human disturbance is probably too broad a generalisation to be useful for predicting invasion success.

(ix) Give birth to live young, are mouth brooders or exhibit parental care of eggs or young

Giving birth to live young (eg guppies), being a mouth brooder (eg *Gambusia* spp), or exhibiting parental care of eggs or young, may enhance survival and hence increase the probability of establishment (Arthington et al 1999, Kailola 2000, Elvira 2001).

Guarding of free-swimming young, as seen in cichlids and the walking catfish, may enhance survival over that of native species with less advanced or no parental care and hence promote establishment success (Taylor et al 1984, Arthington et al 1999, Nico and Fuller 1999). Only anecdotal evidence was found to support this theory. For example, Taylor et al (1984) suggested that spotted tilapia (*Tilapia mariae*), black acaras (*Cichlasoma bimaculatum*) and firemouth cichlids (*Cichlasoma meeki)* are less prone to nest desertion, and hence egg loss to predators, than native fish species in Florida.

Risk assessment significance: A link between establishment success and live births and/or parental care is too uncertain for these factors to be of use for quantitative risk assessment.

(x) Fertilised female able to colonise alone

Some ecologists suggest that vertebrates in which the fertilised female is able to colonise alone should be more successful than those in which the female alone is unable to colonise (Ehrlich 1989). If a solitary pregnant female can found a population, this may increase the number of opportunities for establishment that occur compared to species that require a larger founding group.

No quantitative evidence could be found to support this theory.

Risk assessment significance: Given the lack of evidence supporting this theory, and lack of knowledge about which species would meet this criterion, this factor is unlikely to be of use for assessing the risk of new species establishing.

(xi) Piscivore and detritivore/omnivore dietary groups introduced to low-disturbance habitats

From their observations of Californian fish invasions, Moyle and Light (1996ab) concluded that piscivores (fish eaters) and detritivore/ omnivores are more likely to succeed in establishing in systems with low levels of human disturbance, than fish from other dietary groups. They suggested this relationship also appeared to be true for fish invasions of other freshwater ecosystems. Moyle and Light (1996b) suggested that the success of these two trophic groups is related to the high availability of food during the establishment phase of invasion. They suggested that piscivores and detritivore–omnivores use foods 'that rarely seem to be limiting in aquatic systems'.

Other than Moyle and Light's (1996b) observations, no evidence was found to support this theory.

Risk assessment significance: Given the lack of quantitative evidence supporting this theory, and lack of knowledge and difficulty of classifying dietary groups into clear categories, it would be difficult to apply this theory to risk assessment. Further, many future releases of exotic fish species are likely to occur in systems with high levels of human disturbance, where this factor does not apply, so it is probably of little use for risk assessment.

(xii) Zooplanktivores introduced to lakes

From their observations of Californian fish invasions, Moyle and Light (1996b) concluded that zooplanktivores have a high success rate when introduced to lakes. They suggested that this success is due to the high availability of zooplankton in lakes, ensuring that food is not limiting during the invasion stage.

Other than Moyle and Light's (1996b) observations, no evidence was found to support this theory.

Risk assessment significance: Given the lack of quantitative evidence supporting this theory, this factor is probably of little use for risk assessment.

(xiii) Individual's age and health

A breeding group of fit, healthy young animals would have a better chance than one of less healthy or older animals approaching the end of their reproductive lifespan. The health (including disease status, parasite loading and any stress or debility associated with being kept in captivity) and age (including reproductive lifestage and sufficient lifespan to outlive unfavourable conditions) of the individual animals released may affect establishment chances.

Kailola (2000) speculated that if an introduced species is free of its natural diseases and parasites when it is introduced, this healthy condition may give it a competitive advantage and so enhance its invasive ability. Kailola (2000) presented examples of introduced fish populations that have fewer parasites or diseases than populations of the same species in their endemic range. However, no evidence was found to support the theory that this condition gives such fish enhanced invasion success.

Risk assessment significance: Given future releases of exotic species are likely to be unintentional or illegal, managers are likely to have little opportunity to affect the age or health of released animals, so these variables are unlikely to be of use for managing the risk of new species establishing.

(xiv) Aggressive behaviour and territoriality

Fish that are very aggressive may eliminate native fish through a combination of predation and competition, and so be able to usurp the resources previously used by these native species (Moyle 1986). Territorial behavior may be linked to aggressive behaviour (Arthington et al 1999).

No quantitative evidence was found to support this theory.

Risk assessment significance: This factor has unknown predictive value, so it is not of value for risk assessment.

(xv) Gregariousness

Some ecologists suggest that gregarious fish may be more successful than solitary ones at establishing exotic populations (Ricciardi and Rasmussen 1998, Arthington et al 1999, Elvira 2001). If they are released in a group, fish that form schools or breeding colonies may be more successful invaders because this behaviour facilitates breeding when numbers are low and may also provide protection from predators, make foraging more efficient, and make water temperature more hospitable (Arthington et al 1999).

No evidence or analyses were found that tested this theory for fish.

Risk assessment significance: There is no evidence that evaluations of gregarious behaviour will assist in predicting establishment success.

(xvi) Body size

Animals with larger body size may be more successful at establishing exotic populations than smaller, related species (Nico and Fuller 1999, Kailola 2000, Duncan et al 2001).

Fish with a medium body size or larger may have an advantage, because they are less likely to be preyed on and so may possess enhanced competitive ability. Longevity and fecundity also increase with body size, increasing a species' ability to rapidly increase population size and range. Further, bigger fish tend to exhibit less variation in population size. Both the latter two factors will reduce the extinction risks associated with populations of smaller fish (Townsend 1996).

According to Nico and Fuller (1999), non-indigenous fish in Florida that are most widespread and common are those of medium body size or larger. However, a comparison of the 31 species of exotic fish established in Australia with the 19 species introduced but not established indicates that there is no difference in the mean maximum body size of the two groups (Bomford and Glover 2004).

Risk assessment significance: Body size is unlikely to have any value for predicting the probability of establishment success.

(xvii) Source of animals

Wild-caught animals are more successful at establishing exotic populations than captive-reared animals (Griffith et al 1989, Wiley et al 1992, Snyder et al 1994, Wolf et al 1996). Wild-caught animals may have better skills in avoiding predators and seeking out mates, food and other resources needed for survival and breeding.

No evidence or analyses were found that tested this theory for fish.

Risk assessment significance: This factor has unknown predictive value for fish so it is not of value for risk assessment.

(xviii) Public and government attitudes and actions

Attempts to feed or shelter released animals might increase their chances of establishment. Conversely, attempts to recapture or destroy released animals may reduce their chances of establishment (Bomford 1991). Attempts to feed or shelter released animals may be more likely to occur for attractive or valuable animals and might assist establishment by providing favourable 'microhabitats'. Attempts to recapture or destroy released animals may help prevent establishment and are probably more likely to occur if government policies and practices support them.

No evidence was found that care following release increases establishment success for exotic fish. Government actions to eradicate newly established populations have sometimes been successful. However, such attempts may also fail.

Risk assessment significance: It is uncertain if dedicated assistance can help to establish populations. Attempts to capture or destroy released animals or their progeny may help to reduce the chance of establishment. Public education programs may reduce the chances of exotic fish being released (Rahel 2007).

4.2 Risk assessment for the establishment of exotic freshwater fish introduced to Australia

Exotic freshwater and estuarine fish species have a world introduction success rate of 72.4% (Bomford et al unpublished data, Arthington et al 1999).

<u>Exotic Freshwater Fish Model 1</u> (Bomford and Glover 2004, Bomford 2006):

Bomford and Glover (2004) and Bomford (2006) developed a model for freshwater and estuarine species based on five risk factors and this model gave good predictions for the 49 (31 successful and 18 failed) species known to have been introduced to Australia then (Figure 4.1). Establishment Risk Scores from this model for exotic freshwater fish introduced to Australia are calculated from the sum of five individual risk scores:

1. Climate Match Score
2. Overseas Range Score
3. Establishment Score
4. Introduction Success Score
5. Taxa Risk Score.

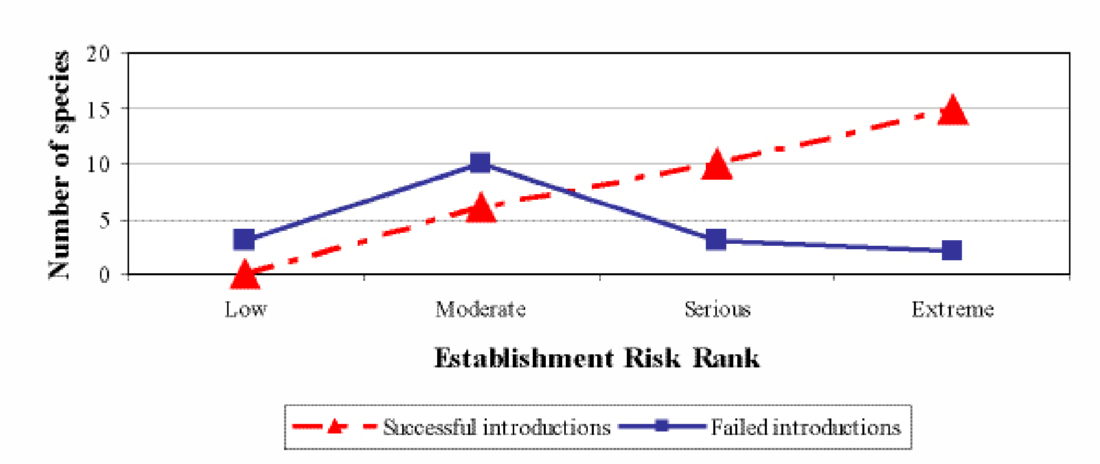

Figure 4.1 Number of species in each Establishment Risk Rank compared for successful and failed exotic freshwater finfish introduced to Australia
Establishment Risk Scores and Ranks were calculated using the directions given in Section 4.3.

Instructions for calculating Establishment Risk Scores and Ranks are presented in Section 4.3. Establishment Risk Scores and Establishment Risk Ranks for exotic fish introduced to Australia are presented in Appendix D, Table D1. The numbers of species in each Establishment Risk Rank are presented in Figure 4.1. This figure shows that the Establishment Risk Ranks for exotic fish introduced to Australia strongly predict introduction outcomes.

<u>Exotic Freshwater Fish Model 2</u> (Bomford et al unpublished):

Since Bomford and Glover (2004) and Bomford (2006) developed the above model, two of the failed species have established in Australia: rosy barb (*Puntius conchonius*) and pearl cichlid (*Geophagus brasiliensis*). These successful establishment events meant the sample size of failed species was too small to enable robust quantitative tests of the factors influencing establishment outcomes.

Therefore, Bomford et al (unpublished) developed a generalised linear mixed model to describe probability of establishment success for fish introduced to ten countries (including Australia) to determine factors affecting introduction outcomes. As these authors found that jurisdiction had no significant effect on establishment success for exotic freshwater fish, their model for these ten countries can be applied to Australia. A predictive model was fitted using only those terms determined to have a significant effect on establishment success. The generic model is based on Climate 6 (as opposed to Climate 5, 7 or 8), since Climate 6 was shown to be the best predictor of success of introduction. Country of introduction and success rate of species in the same genus were not included as factors in the model, as they were not significant predictors of success of introduction. The purpose of this model is to allow a risk assessment to be conducted prior to the attempted introduction of a fish species.

Bomford et al's (unpublished data) model for the probability of establishment of exotic freshwater fish is:

P(Establishment) = exp(-3.2974 + 2.9611(*prop.species*) + 3.2948(*prop.family*) + *s (Climate 6) + Family random effect*))

P(Establishment) = probability of establishment

Prop.species = number of countries where species successfully established divided by the total number of countries where species introduced

Prop.family = number of successful introductions to all countries of species in the family divided by the total number of introductions to all countries of species in the family

S(Climate 6) = a smooth function of the climate match score expressed as a proportion of all data locations in the jurisdiction. Instructions for calculating this variable are presented in Section 4.4.

Family random effect = a family random effect assumed drawn from a Gaussian distribution with mean zero and variance that was estimated from Bomford et al's (unpublished) data. A table listing these values is presented in Section 4.4.

Instructions for using Bomford et al's (unpublished) model for ranking establishment risk for species introduced to Australia are presented in Section 4.4. This model can only be used for species that have been introduced to at least three countries so that success rates can be calculated (Section 4.4), and for fish in families that were included in developing this model (Section 4.4). Results for the species introduced to these countries are presented in Appendix D, Table D3.

P(Establishment) values for exotic fish introduced to ten countries calculated using Bomford et al's (unpublished data) model are presented in Appendix D, Table D2.

P(Establishment) values are converted to Establishment Risk Ranks ranging from Low to Extreme for each species (Section 4.4). Figure 4.2 presents the number of fish species in each Establishment Risk Rank for all species introduced to the ten countries used in Bomford et al's (unpublished data) model. This figure shows that these Establishment Risk Ranks strongly predict introduction outcomes.

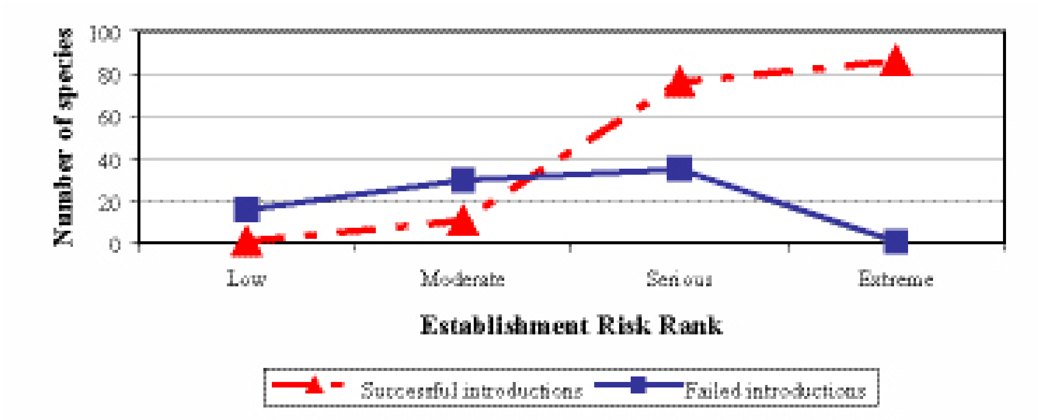

Figure 4.2 Number of species in each Establishment Risk Rank for fish introduced to ten countries

P(Establishment) values were calculated using the model developed by Bomford et al (unpublished data) according to the directions given in Section 4.4 of this report.

Bomford and Glover (2004) and Bomford et al (unpublished data) used Fishbase occurrence records for world geographic ranges to calculate climate matches. Occurrence records in Fishbase do not represent the full world range of many fish species. If the missing range includes climates that are not covered by the included range, analysis may lead to the species' climate match in the target country being underestimated. Should better data be available on a species' world range, its use will give more accurate results. Further development of Bomford and Glover's (2004) and Bomford et al's (unpublished data) models, using more accurate data on species' world ranges, is desirable to potentially improve their predictive accuracy.

Both Exotic Freshwater Fish Model 1 and 2 only assess the likelihood that a species will establish an exotic population, not whether it is likely to spread following establishment. Factors affecting spread may differ from those affecting establishment, although a high climate match will indicate a high potential for spread.

4.3 Instructions for using Exotic Freshwater Fish Model 1

The model for which instructions are presented in this section is the original model published by Bomford and Glover (2004), modified by Bomford (2006) to (i) give a four-rank risk outcome instead of the original six-rank outcome and (ii) use the PC version of CLIMATE rather than the Apple Macintosh version. The CLIMATCH software developed by the Bureau of Rural Sciences can also be used with this model. The model applies to the Australian mainland and Tasmania but not to small offshore islands.

A. Climate Match Score (0–8)

Climate Match Scores are calculated using the table below. For the selected fish species, use PC CLIMATE (Bureau of Rural Sciences 2006) or CLIMATCH (Bureau of Rural Sciences; see http://www.brs.gov.au/climatch) and select:

- *'worlddata_all.txt'* as the world data location
- all 16 climatic parameters for matching locations (see Table 1)
- *'Euclidian match'* for the analysis.

Sum the values for the six highest match classes (ie the scores for match levels 10, 9, 8, 7, 6 and 5) = 'Value X'

Convert Value X to a Climate Match Score (1–8) using the following cut-off thresholds:

CLIMATE Euclidian Sum Level 5 (Value X)	Climate Match Score
0	1
1–40	2
41–150	3
151–400	4
401–1000	5
1001–1500	6
1501–2500	7
> 2500	8

If the input area has 12 or fewer meteorological stations, then it is likely to underestimate the climate match to Australia. If this is the case, it is advisable to increase the climate match score by one increment. For example, if the input range for a species included only five meteorological stations, and the sum of the values for the six highest match classes to Australia equalled 104 (ie Value X = 104), then this would give a Climate Match Score = 3 + 1 = 4.

B. Overseas Range Score (0–4)

Overseas Range Scores are calculated using the table below. Count the number of grid squares (1° latitude x 1° longitude) in which an occurrence of the species is recorded in Fishbase, excluding Australia.

Number of grid squares with species present	Overseas Range Score
≤ 4	0
5–10	1
11–20	2
21–30	3
≥ 31	4

C. Establishment Score (0–3)

Establishment Scores are calculated using the table below. Check Fishbase for locations where successful introductions of the species have occurred, excluding Australia. A moderate risk rank score of 1 is given where there are no recorded introductions, although a precautionary approach could warrant a higher risk score.

Introduction outcome overseas	Establishment Score
Introduced but never established	0
Never introduced	1
Only established exotic population(s) on island(s) or on one continent (from choice of five continents excluding Australia: Africa, Europe, Asia, North and Central America, or South America)	2
Established exotic populations on more than one continent (excluding Australia)	3

D. Introduction Success Score (0–4)

Introduction Success Scores are calculated using the table below. Count the number of known successful introductions of the species worldwide excluding Australia and express this as a proportion of the total number of introductions. A moderate Introduction Success Score of 2 is given where there are no recorded introductions, although a precautionary approach could warrant a higher Introduction Success Score.

Introduction success rate	Introduction Success Score
0	0
>0–0.25	1
>0.25–0.5	
OR	
Never introduced	2
>0.5–0.75	3
>0.75–1.0	4

E. Taxa Risk Score (0–5)

The Taxa Risk Scores are success rates for worldwide introductions of the family or genus of the species being assessed. The Taxa Risk Score is either a species' Genus Risk Score, or where there are too few introduction records within the species' genus to enable a Genus Risk Score to be calculated, an alternative Family Risk Score is calculated, using the tables below.

Genus Risk Score

The Genus Risk Score is used as the Taxa Risk Score when the number of introduction events of all species within the same genus as the species being assessed is greater than or equal to 4.

The Genus Risk Score is calculated from all recorded worldwide introductions of all species within the same genus as the species being assessed.

Genus success rate % = 100(Number of successful introductions to all countries of species in the genus/Total number of introductions to all countries of species in the genus)

For example, if eight species from the genus *Demo* were introduced to United Kingdom and three established, and the only other introduction from this genus was two species introduced to Japan, of which only one was successful, this would give a Genus success rate % for *Demo* of 100 × (4/10) = 40%.

Genus success rate (%)	**Genus Risk Score**
0	0
>0<10	1
10–25	2
>25<40	3
40–60	4
>60	5

Family Risk Score

The Family Risk Score is used as the Taxa Risk Score to increase the sample size when number of introduction events of all species within the same genus as the species being assessed is between 0 and 3.

The Family Risk Score is calculated from all recorded worldwide introductions of all species within the same family as the species being assessed:

Family success rate % = 100(Number of successful introductions of species to all countries in the family/Total number of introductions to all countries of species in the family)

For example, if five species in the Family Exampleidae were introduced to United Kingdom and three established, and the only other introduction from this family was one of the same species introduced to Japan that failed, this would give a score of 100 x (3/6) = 50%.

Where there are no recorded introductions, or where sample sizes are small, a moderate (or more moderate) Family Risk Score is given, although a precautionary approach could warrant a higher Family Risk Score.

Family success rate (%)	Family Risk Score
0 (number introductions ≥ 3)	0
0 (number introductions 1–2)	1
1–25 (any number introductions)	2
OR	
Never introduced (number introductions 0)	
>25–60 (any number introductions)	3
>60 (number introductions 1–2)	4
>60 (number introductions ≥ 3)	5

Establishment Risk Score

An exotic finfish species' Establishment Risk Score is calculated from the sum of its five scores for A to E above.

Establishment Risk Rank

An exotic finfish species' Establishment Risk Score is converted to an Establishment Risk Rank ranging from Low to Extreme using the following cut-off thresholds:

Establishment Risk Rank	Establishment Risk Score
Low	≤ 7
Moderate	8–14
Serious	15–19
Extreme	≥ 20

4.4 Instructions for using Exotic Freshwater Fish Model 2 to rank establishment risk for fish introduced to Australia

The model for which instructions are presented in this section is based on analyses by Bomford et al (unpublished data) for exotic fish introduced to ten countries. Hence, some parameter values required for using the model are only available for the taxa that had been introduced to these countries. The model applies to the Australian mainland and Tasmania but not to small offshore islands.

A. Family Random Effect value

Family Random Effect Values are only available for the families of species that were used in Bomford et al's (unpublished data) analysis of species that had been introduced to ten countries. These values are presented in Table 4.2.

Table 4.2 Family Random Effect Values for exotic fish introduced to ten countries.

Family	*Family Random Effect*	Family	*Family Random Effect*
Acipenseridae	-0.008	Esocidae	-0.019
Anguillidae	-0.012	Gasterosteidae	0.0008
Atherinidae	0.008	Gobiidae	0.009
Belontiidae	0.007	Ictaluridae	0.040
Catostomidae	-0.005	Moronidae	0.057
Centrarchidae	-0.045	Osphronemidae	-0.007
Centropomidae	-0.016	Osteoglossidae	-0.009
Characidae	0.012	Percidae	0.013
Cichlidae	-0.040	Poeciliidae	0.004
Clariidae	-0.062	Salmonidae	-0.085
Cobitidae	0.004	Siluridae	0.002
Cyprinidae	0.13	Umbridae	0.024

B. Prop.species value

Prop.species value = number of countries where species successfully established/total number of countries where species has been introduced.

Table 4.3 presents *Prop.species* values for fish calculated for those species that have already been introduced to at least three countries outside their native range, from data taken from Arthington et al's (1999) global database of exotic fish.

Table 4.3 *Prop.species* values for fish for which there are world records for introductions to three or more countries worldwide.

Species	*Prop.species*	Species	*Prop.species*
Acipenser baerii	0.14	*Catla catla*	0.50
Acipenser gueldenstaedtii	0.00	*Channa striata*	0.75
Ambloplites rupestris	1.00	*Cichla ocellaris*	1.00
Ameiurus melas	1.00	*Cichlasoma facetum*	1.00
Ameiurus nebulosus	0.95	*Cichlasoma meeki*	1.00
Anabas testudineus	0.75	*Cichlasoma nigrofasciatum*	0.67
Anguilla anguilla	0.44	*Clarias batrachus*	0.80
Anguilla japonica	0.17	*Clarias gariepinus*	0.57
Aristichthys nobilis	0.30	*Colossoma macropomum*	0.00
Astatoreochromis alluaudi	0.75	*Coregonus clupeaformis*	0.00
Astronotus ocellatus	0.67	*Coregonus lavaretus*	0.75
Barbodes gonionotus	0.86	*Coregonus peled*	0.83
Betta splendens	0.80	*Ctenopharyngodon idella*	0.55
Carassius auratus	0.98	*Cyprinus carpio*	0.91
Carassius carassius	0.88	*Esox lucius*	0.89
Carpiodes cyprinus	1.00	*Gambusia affinis*	0.96

Species	Prop.species
Gambusia holbrooki	1.00
Gobio gobio gobio	1.00
Hemichromis bimaculatus	1.00
Hemiculter leucisculus	1.00
Heterotis niloticus	1.00
Hoplias malabaricus	0.33
Hucho hucho	1.00
Hypophthalmichthy molitrix	0.68
Ictalurus punctatus	0.40
Ictiobus bubalus	0.67
Ictiobus cyprinella	0.17
Ictiobus niger	0.33
Labeo rohita	0.22
Lates niloticus	0.60
Lepomis auritus	1.00
Lepomis cyanellus	0.63
Lepomis gibbosus	1.00
Lepomis macrochirus	0.86
Lepomis microlophus	0.67
Leucaspius delineatus	1.00
Leuciscus idus	0.67
Limnothrissa miodon	1.00
Megalobrama terminalis	0.00
Micropterus dolomieu	0.29
Micropterus punctulatus	0.67
Micropterus salmoides	0.83
Misgurnus anguillicaudatus	1.00
Morone chrysops	0.25
Morone saxatilis	0.33
Mylopharyngodon piceus	0.38
Neogobius melanostomus	1.00
Odontesthes bonariensis	1.00
Oncorhynchus clarki clarki	0.00
Oncorhynchus gorbuscha	0.67
Oncorhynchus keta	0.00
Oncorhynchus kisutch	0.30
Oncorhynchus mykiss	0.74
Oncorhynchus nerka	0.17
Oncorhynchus rhodurus	0.00
Oncorhynchus tshawytscha	0.25
Oreochromis aureus	0.85
Oreochromis hornorum	0.90

Species	Prop.species
Oreochromis macrochir	0.33
Oreochromis mossambicus	0.94
Oreochromis niloticus	0.86
Oreochromis spirulus	0.75
Oryzias latipes	0.67
Osphronemus goramy	0.33
Perca fluviatilis	1.00
Phalloceros caudimaculatus	1.00
Pimephales promelas	0.75
Poecilia latipinna	0.90
Poecilia mexicana	1.00
Poecilia reticulata	0.97
Polyodon spathula	0.00
Pomoxis annularis	0.67
Pomoxis nigromaculatus	0.60
Pseudorasbora parva	1.00
Puntius conchonius	0.80
Rhodeus sericeus	1.00
Rutilus rutilus	0.80
Salmo salar	0.33
Salmo trutta	0.76
Salvelinus alpinus	0.38
Salvelinus fontinalis	0.65
Salvelinus namaycush	0.64
Sander lucioperca	1.00
Sarotherodon galilaeus	0.57
Sarotherodon melanotheron	0.67
Scardinius erythrophthalmus	1.00
Serranochromis robustus	1.00
Silurus glanis	1.00
Tilapia rendalli	0.81
Tilapia zillii	0.89
Tinca tinca	0.84
Trichogaster lalius	0.67
Trichogaster leerii	0.67
Trichogaster pectoralis	0.89
Trichogaster trichopterus	0.71
Tridentiger trigonocephalus	1.00
Umbra pygmaea	1.00
Xiphophorus hellerii	0.93
Xiphophorus maculatus	0.89
Xiphophorus variatus	0.67

Values were calculated from Arthington et al’s (1999) global database of exotic fish.

C. Prop.family value

Prop.family value = Number of successful introductions of species to all countries in the family/ Total number of introductions to all countries of species in the family.

For example, if five species in the family Exampleidae were introduced to United Kingdom and three established, and the only other introduction from this family was one of these species that was also introduced to Japan and failed to establish, this would give a score of 3/6 = 0.5.

Table 4.4 presents *Prop.family* values for fish calculated for those families which have already been introduced to at least three countries outside their native range from data taken from Arthington et al's (1999) global database of exotic fish.

Table 4.4 Prop.family values for fish for which there are world records for introductions to three or more countries worldwide

Family	*Prop.family*	Family	*Prop.family*
Acipenseridae	0.05	Erythrinidae	0.33
Adrianichthyidae	0.67	Esocidae	0.75
Anabantidae	0.67	Fundulidae	0.75
Anguillidae	0.31	Gasterosteidae	1.00
Aplocheilidae	0.67	Gobiidae	1.00
Atherinidae	1.00	Ictaluridae	0.84
Belontiidae	0.81	Moronidae	0.36
Catostomidae	0.44	Osmeridae	1.00
Centrarchidae	0.77	Osphronemidae	0.33
Centropomidae	0.50	Osteoglossidae	0.75
Channidae	0.82	Percidae	0.96
Characidae	0.25	Poeciliidae	0.93
Cichlidae	0.84	Polyodontidae	0.00
Clariidae	0.68	Salmonidae	0.55
Clupeidae	0.89	Siluridae	1.00
Cobitidae	1.00	Umbridae	0.71
Cyprinidae	0.73		

Values were calculated from Arthington et al's (1999) global database of exotic fish.

D. *S(Climate 6)* value

Use PC CLIMATE (Bureau of Rural Sciences 2006) or CLIMATCH (Bureau of Rural Sciences; see http://www.brs.gov.au/climatch) and select:

- *'worlddata_all.txt'* as the world data location
- all 16 climatic parameters for matching locations (see Table 1)
- *'Euclidian match'* for the analysis
- Splined (gridded) surface for Australian 'match to' file.

Perform a Euclidian match and then calculate the sum of the five scores for classes 6 to10. Express this as a proportion of the maximum possible score (that is 2785 for Australia).

Look up the Climate 6 score along the x-axis of Figure 4.3. Read off the y-axis the equivalent *S(Climate 6)* value.

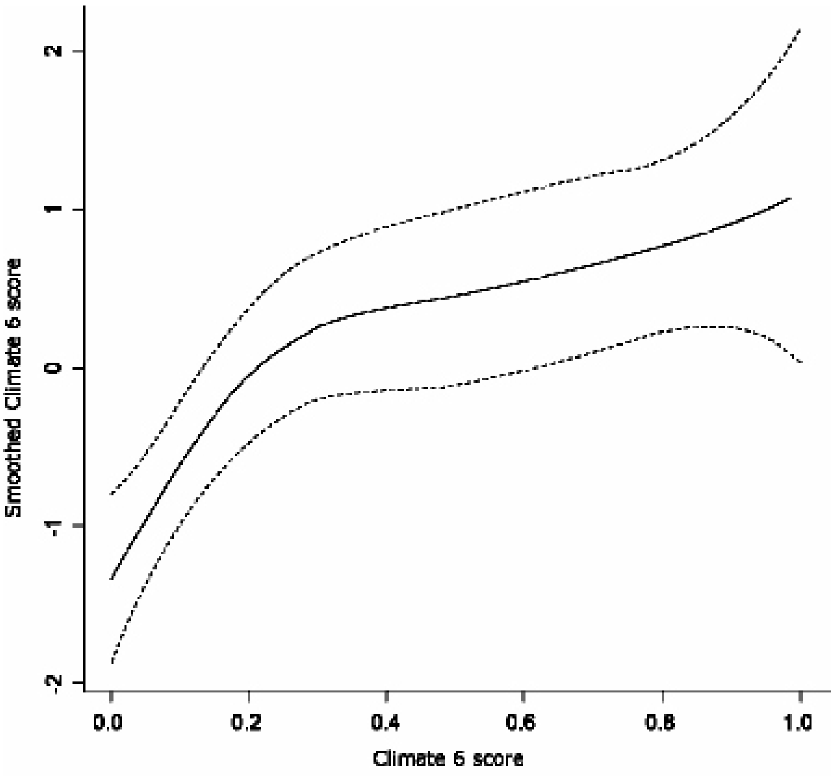

Figure 4.3 Penalised regression spline fit of Climate 6 score for freshwater fish
The solid line (fitted by the model) indicates that as the Climate 6 score increases (x-axis) the chance of successful introduction increases. The dotted lines indicate the 95% confidence interval around the line. The solid line is used to convert raw Climate scores to Smoothed Climate scores.

For example, the carp (*Cyprinus carpio*) scores:

Euclidian match scores sum levels 6 to10 = 1679

Climate 6 score for the carp = 1679/2785 = 0.603

S(Climate 6) value for the carp from Figure 4.3 = 0.60.

P(Establishment)

P(Establishment) = 1/(1 + exp(3.2974 – 2.9611(*prop.species*) – 3.2948(*prop.family)* – *s (Climate 6)* – *Family random effect*))

For example, the carp (*C. carpio*) is in the family Cyprinidae. P(Establishment) for carp for Australia

= 1/(1 + exp (3.2974 – 2.9611(0.91) – 3.2948(0.73) – 0.60 – 0.13))

= 1/(1 + exp(3.2974 – 2.6946 – 2.4052 – 0.60 0.13))

= 1/(1 + exp(-2.53))

= 1/(1 + 0.080)

= 0.93.

Establishment Risk Rank

P(Establishment) values are converted to Establishment Risk Ranks using the following conversions:

Establishment Risk Rank	*P(Establishment)*
Low	≤0.12
Moderate	0.13–0.40
Serious	0.41–0.89
Extreme	≥ 0.90

For example, the P(Establishment) for carp introduced to Australia is 0.93, which is greater than 0.90, giving carp an Establishment Risk Rank of Extreme.

Other species

Family Random Effect values are small, relative to the other parameters in the model, which means they do not have a big effect on the *P(Establishment)* values. For species whose Family Random Effect values are not included in Table 4.2, potential minimum and maximum *P(Establishment)* values could be calculated by inserting the minimum (-0.085) and maximum (0.13) Family Random Effect values from Table 4.2 into the model. These minimum and maximum *P(Establishment)* values could then be used to calculate the minimum and maximum Establishment Risk Rank(s) for the species.

Prop.species and *Prop.family* values in Tables 4.3 and 4.4 were calculated from introduction records in Arthington et al's (1999) database. Only species and families for which there were three or more introduction records are included in these tables. For species not included in these tables, data on successful and failed introduction records would need to be obtained from other sources. Arthington et al's (1999) database includes many species' introduction records for which the outcome (succeeded or failed to establish) is unknown or uncertain. These records were excluded from Bomford et al's (unpublished data) analyses and from Tables 4.3 and 4.4. A check of more recent databases, such as Fishbase (2008), might confirm the outcome of these introductions and so enable *Prop.species* and *Prop.family* values to be calculated for some of these missing taxa.

4.5 Factors affecting assessment of pest status of introduced freshwater fish

Fishbase (2008) lists 32 species of exotic freshwater fish that have been reported by three or more countries as having adverse ecological impacts. Most of these species have a High–Extreme risk of establishing in Australia (Bomford and Glover 2004). Many other fish species could pose similar risks. Factors affecting the assessment of the potential pest status of exotic fish are described below.

4.5.1 Reliability of evidence

Unfortunately, for most exotic finfish, both in Australia and overseas, reliable knowledge about impacts is sparse for two main reasons. First, there has been limited research and in particular there are usually scarce preinvasion data sets (Ojaveer et al 2002). Secondly, introductions of exotic finfish have often coincided with other changes to freshwater and estuarine habitats which means impacts due to exotic fish are confounded with impacts due to other factors (McKay 1984; Moyle 1986; Welcomme 1988; Moyle and Williams 1990; Crowl et al 1992; Arthington and Mckenzie 1997; Kailola 2000; Elvira 2001; Ojaveer et al 2002; Rahel 2000, 2002; Cambray 2003; Dextrase and Mandrak 2006). These factors include the following:

- changed water flows — for example, due to weirs, dams, irrigation or channel straightening
- reduced water quality — including chemical pollution, modified temperature regimes, turbidity and reduced oxygen
- fishing — including collections for aquaria
- introductions of exotic plants
- disturbance by other introduced animals and people — for example, grazing or cropping in water catchments or along banks, clearing of snags and logs.

These factors are often cumulative or complementary and may interact synergistically, such that the impact of several factors acting together is greater than the sum of the individual factors acting alone (Elvira 2001). For example, some native fish might survive predation by introduced fish unless habitat disturbance destroys aquatic plants they use for shelter, so they are unable to hide from the predatory fish. Such interactions make it difficult to accurately assign individual causes to specific impacts.

The combined effects of introduced species and human-caused environmental changes may cause rapid and unpredictable changes in fish assemblages (Herbold and Moyle 1986, Meng et al 1994). For example, in New Zealand, deforestation and swamp drainage have had detrimental impacts on native species. In areas of New Zealand where human population is low, and where deforestation and land development are less than elsewhere, fish such as galaxiid stocks and retropinnid smelts remain productive or abundant in spite of the co-occurrence of introduced trout (McDowall 1990).

Many of the impacts attributed to exotic fish are correlative or anecdotal (King 1995). Nonetheless, the diet and behaviour of some finfish definitely give them the potential to harm native fish and cause other environmental damage in their introduced habitats. This potential, combined with measured changes in abundance or distribution of vulnerable native species following their introduction to new habitats, provides compelling evidence of harmful impacts (Moyle 1986). For example, Yang (1996) recorded that 18 exotic fish species have been introduced in Yunnan Province in China and a further 16 species that were not originally present in Yunnan have been translocated from elsewhere in China. Yunnan had 432 documented freshwater fish species, but following the introduction of the exotic and translocated fish many of these native fish have declined or disappeared: 130 of the endemic fish have not been caught for the last five years, a further 150 species that were common are now rare and the remaining 152 species have declined. The introduced fish affect the endemic fish directly by eating their spawn and competing for food. They also impact indirectly, by encouraging changed fishing methods that have a greater impact on the native species than previous methods. Although other disturbances have occurred in these habitats, including land reclamation, irrigation works and overfishing, an analysis of the timing of endemic fish declines in relation to the timing of exotic fish introductions and other disturbances indicated that the introduced fish were the main factor causing declines in the endemic fish (Yang 1996).

4.5.2 State of knowledge on impacts

It is not possible to estimate a reliable figure for the percentage of exotic fish that become pests, because few reliable data on fish impacts are available. Hence, impacts due to exotic fish are largely under-reported in the scientific literature. However, several studies have attempted to estimate the proportion of exotic fish that have detrimental environmental impacts:

- Maciolek (1984) reviewed fish introductions to Pacific Ocean islands and found 14 of 31 (45%) introduced fish species had substantial impacts on native fauna, either directly or indirectly.
- Welcomme (1988) examined FAO records of 1354 introductions of 237 exotic fish species into 140 countries between 1800 and 1985. He found the introduced species were considered a significant element in their new habitat in 23.7% of introductions but were only considered to be a serious environmental pest in 6.6% of introductions.
- Ross (1991) examined 31 studies of the introduction of exotic fish to 26 aquatic systems in Europe, North America, Australia and New Zealand. 77% of these studies reported a decline in native fish numbers following the introduction of exotic or translocated fish. Of the 26 systems studied, 20 reported native fish declines, eight attributed the declines to predation and eight to competition, with no mechanism identified for the other four systems.

Given that the impacts of most introductions will not have been studied, the figures above could be significant underestimates of true impacts. Cassey and Arthington (1999) suggested the low percentage of fish considered to be pests is most likely an artefact of the scale of most studies. They also suggested that most changes will be subtle effects, such as local extinctions, behavioural and evolutionary changes of native species, habitat and environment changes, food web alterations, and transmission of pathogens. Such effects are rarely investigated in detail (Townsend 1991).

4.5.3 Types of environmental impact and their significance for impact risk assessment

A review of the literature on exotic finfish introductions indicates a variety of impacts may occur. These impacts are briefly described below, together with examples and their risk assessment significance.

(i) Competition for resources

Competition can lead to reduced growth rates, survival and recruitment (Taylor et al 1984; Welcomme 1988; Arthington and Lloyd 1989; Arthington 1989, 1991; Ross 1991; Crowl et al 1992; Lever 1996; Moyle and Light 1996ab; Kailola 2000; Ojaveer et al 2002). Two types of competition may occur: exploitation and interference competition (Pianka 1978).

Exploitation competition occurs when different species use common resources that are in short supply; most commonly food and space. This competition may lead to displacement of the weaker species to less favourable foods and habitats (niche shifts) and hence cause reduced survival and recruitment (Ross 1991). For example, McIntosh et al (1992) found that the native fish *Galaxias vulgaris* occurs at much lower densities in the presence of introduced brown trout (*Salmo trutta*) in New Zealand. Both taxa exhibit considerable dietary overlap, and most competition centres around optimal feeding locations. Similarly in Australia, the diets of *G. olidus* and *S. trutta* overlap and where the species occur in the same waterway, the distribution of the galaxiids is fragmented through interspecific competition for food (Fletcher 1979).

Interference competition occurs when different species seeking a common and abundant resource harm each other in the process (eg by aggressive behaviour). For example, mosquitofish (*G. holbrooki*) in Australia compete with native species for resources, by fin nipping and aggression towards native fish up to twice their size — this behaviour can reduce survival and recruitment of the attacked species (Arthington and Lloyd 1989).

Exotic fish are often better adapted to disturbed habitats than native fish, which can enhance their competitive advantage in these habitats (Moyle and Light 1996b, Arthington et al 1999). Arthington (1989, 1991) reviews the impacts of competition between exotic and native freshwater fish in Australia. She suggested that because many exotic fish are generalist feeders that exhibit trophic opportunism (diet flexibility dependent on available food), there is considerable potential for competition between native and exotic fish, and there is evidence that such competition has occurred and caused declines in some native species.

Moyle and Light (1996b) suggested that when exotic fish invade constant environments, such as desert springs, tropical lakes or artificial reservoirs, they often have highly adverse effects on native species, because in an unvarying environment it is much easier for a single species or group of species to become dominant. In contrast, in a fluctuating environment with varying resource types and availability, no one species or group of species can stay dominant for an extended period.

In general, there has been insufficient research to determine the extent to which competition from exotic fish has detrimentally affected native fish. According to Herbold and Moyle (1986) and Moyle et al (1986), introduced fish do not fill 'vacant niches'. Rather, they compress the realised niches of one or more of the species already present, possibly to the point where pre-existing species are eliminated.

Risk assessment significance: Competition by exotic fish has the potential to be highly detrimental to native species. However, scientific knowledge is currently inadequate to allow reliable predictions about which exotic species will have the worst impacts when they are introduced to new environments. Elvira (2001) stated that species associated with high impacts tend to have a broad diet, whereas introduced fish having low impacts are characterised by specialised diet. Hence, generalist feeders may have more potential to have competitive impacts than specialist feeders.

(ii) Predation

Predation leads to reduced survival rates of prey species (Taylor et al 1984; Arthington 1989, 1991; Crowl et al 1992; Lever 1996; Moyle and Light 1996ab; Kailola 2000). For example, when predatory Nile perch were introduced to Lake Victoria and other similar lakes in Africa, they caused the loss of many native cichlid species (Welcomme 1988). In Australia, mosquitofish (*G. holbrooki*) attack and eat juvenile fish and may also eat fish fry (Arthington and Lloyd 1989, Kailola 2000). Predation by mosquito fish may contribute to declines in native fish populations in Australia including populations of firetail gudgeon (*Hypseleotris galli*), western minnow (*Galaxias occidentalis*), nightfish (*Bostockia porosa*), westralian pygmy perch (*Edelia vittata*), gudgeons (*Mogurnda* spp), glassy perch (*Ambassis* spp), rainbowfish (*Rhadinocentrus ornatus*, *Melanotaenia fluviatilis* and other *Melanotaenia* spp), blue-eyes (*Pseudomugil* spp), hardyheads (*Craterocephalus* spp) and smelt (*Retropinna* spp) (Arthington et al 1983,1999; Arthington and Lloyd 1989; Lloyd 1990; Ivantsoff and Aarn 1999). Arthington and Marshall

(1999) considered that the capacity of mosquito fish to feed opportunistically on a wide variety of aquatic prey, its consumption of fish eggs and larvae and its aggressiveness towards other fish species, could certainly exert significant pressure on small populations of indigenous fish themselves already under threat from habitat loss and water pollution.

Carnivorous fish may also have detrimental impacts on prey populations of taxa other than fish. For example, where exotic species of trout have been introduced into protected areas of California, populations of the native yellow-legged frog (*Rana muscosa)* and Pacific treefrog (*Hyla regilla*) have declined (Knapp and Matthews 2000, Matthews et al 2001). Brown trout (*Salmo trutta*) introduced to New Zealand, caused a strong trophic cascade, affecting not only populations of native fish such as *Galaxis eldoni* and grazing invertebrates, but also net primary production and the rate of algal photosynthesis (Townsend and Simon 2006). Some cyprinodonts, particularly *Gambusia* spp, may feed on the eggs of other taxa (Welcomme 1988). For example, tadpoles of native frog species in Australia are highly susceptible to predation by *G. holbrooki* and hence this species may contribute to declining frog populations (Morgan and Buttemer 1996, Webb and Joss 1997, Kailola 2000).

According to Crowl et al (1992), many Australian endemic species are likely to have evolved in relative isolation from aggressive predatory fish and hence are particularly prone to negative impacts from introduced predators. For example, Crowl et al (1992) suggested that the family Galaxiidae, found only in the southern hemisphere, have little co-evolutionary history with predators. According to Arthington et al (1999), isolated aquatic communities are particularly at risk from introduced predators, and piscivores may pose an especially high risk. Piscivores may be more likely than fish from other dietary groups to alter invaded communities. Moyle and Light (1996b) considered that a relatively small number of invasive fish species, mainly piscivores, are responsible for most recorded cases of native fish extinctions caused by invading fish. Many of these predatory fish were introduced intentionally for sport fisheries.

Risk assessment significance: Predation by exotic fish has the potential to be highly detrimental to native species, since piscivores may be more likely to alter invaded communities and are known to cause native fish extinctions.

(iii) Habitat disturbance and food web effects

Habitat alterations occur when introduced fish change the habitat of resident species, often through their feeding behaviour (Taylor et al 1984; Arthington 1989, 1991; Crowl et al 1992; Lever 1996; Townsend 1996 and 2003; Kailola 2000; Elvira 2001; Ojaveer et al 2002). The most common effects are the displacement of aquatic vegetation and the degradation of water quality. Fish can remove plants from habitats by eating them or by uprooting them through digging for food or nesting sites (Taylor et al 1984, Elvira 2001). This plant removal can change complex habitats into simple ones (Crowl et al 1992). Grass carp (*Ctenopharyngodon idella*) in Donghu Lake, China, caused the virtual disappearance of submerged macrophytes, resulting in dramatic blooms of planktonic algae (Kottelat and Whitten 1996). These conditions favoured silver carp and bighead carp — native to China but not to Donghu Lake. Increases in the numbers of these two species resulted in most of the 50 endemic fish species in the lake disappearing. The number of benthic invertebrate species also fell from 113 to 26 and zooplankton species fell from 203 to 171 (Kottelat and Whitten 1996).

Reductions in macrophytes can also cause increases in turbidity through wave-mediated erosion and continual mixing of silt previously stabilised by rooted plants. Turbidity can also be caused by bottom-feeding species, such as European carp, agitating shallow littoral zones, and by fish nesting and spawning activities, especially by species that form dense aggregations for breeding (Taylor et al 1984). Increased turbidity can have detrimental effects on native species by disrupting breeding and reducing recruitment, slowing growth, or interfering with normal respiratory and secretory functions (Taylor et al 1984).

Alterations in ecosystem structure can have flow-on effects to oxygen levels, turbidity, and nutrient cycling, and hence change community assemblages. Bottom-feeding fish, such as cyprinids, transfer nutrients from sediment into the water by excretion, which can contribute to formation of algal blooms (King 1995). In contrast, Australian native fish feed largely within the water column and so do not recycle sediment nutrients (Kailola 2000). Introduced species can also become prey for larger fish, thus changing food availability (Taylor et al 1984, Ross 1991).

Large secondary effects can also result from introductions of predatory fish and these flow-on effects are usually hard to predict (Li and Moyle 1981, Townsend 1991). For example, lake and pond ecosystems are strongly influenced by the feeding behaviour and population dynamics of predatory fish such as trout and *Gambusia* spp (Hurlbert et al 1972, Townsend 1996). Top-level predators can reduce the number of grazing fish, zooplanktivores and large grazing invertebrates. These predators can also reduce the extent and efficiency of the grazing, so producing an increase in phytoplankton and even algal blooms. For example, Moyle and Light (1996b) cited many 'well-documented' case histories of 'dramatic effects of piscivores on fish assemblages in lakes and streams'. Moyle and Light (1996b) said that 'the effects of a predator invasion can "cascade" through an entire ecosystem, altering fundamental ecosystem processes'.

In addition to predators' direct effects of fish removal caused by hunting and aggression, they can also influence community structures by altering the balance of interspecific competition (Ross 1991). Predatory fish could hence alter species diversity in the communities where they are introduced (Ross 1991). Exotic predators can profoundly affect the population dynamics of native prey species (Elvira 2001).

Conversely, the presence of exotic fish may significantly increase the amount of prey available to native predators (Taylor et al 1984, Elvira 2001). For example, the introduced round goby (*Neogobius melanostomus*) may cause relaxation of predation pressure on several native prey fish in the Baltic Sea, such as the sand eel (*Ammodytes tobianus*) and sprat (*Sprattus sprattus*), by being more favourable food for most abundant piscivores (Ojaveer et al 2002).

Based on assessment of fish invasions in Californian streams, lakes and estuaries and in wet zone streams of Sri Lanka (Wikramanayake and Moyle 1989), Moyle and Light (1996b) concluded that detritivores and omnivores are less likely to have harmful effects on fish assemblages in invaded freshwater communities than fish from other dietary groups. However, although they may not eliminate native finfish, detritivores and omnivores may still considerably alter ecosystem functioning (Power 1990, Moyle and Light 1996ab) and hence may possibly cause extinctions of lower-order taxa.

Risk assessment significance: The secondary or flow-on effects in food webs are the least studied and most difficult effects of exotic fish introductions to predict. Exotic fish have the potential to have detrimental effects on recipient ecosystems when they alter the habitat of native species. Species that destroy or modify aquatic vegetation or that stir up sediments to increase turbidity possibly have the highest impacts, but introduced piscivores may also significantly alter community structures. Detritivores and omnivores may be less likely to have harmful effects on fish assemblages.

(iv) Potential to cause injuries

The following attributes give fish the potential to cause injury (modified from McKay 1984):

- strong, serrated or venomous spines that lock into position — most freshwater catfishes
- electric organs — for example, electric eels and catfish
- poisonous flesh — for example, fish in the genus *Tetraodon*
- sharp teeth capable of cutting flesh — for example, piranhas (*Serrasalmus* spp).

For example, the electric eel is potentially dangerous to all animals, including aquatic organisms and land-dwelling animals. These eels transmit direction-finding pulses at a frequency of 50 hertz and are capable of producing shocks reaching one ampere and 600 volts. Fish and mammals as large as horses may be paralysed by electric eels (Department of Primary Industries and Fisheries 2004).

Risk assessment significance: Fish that cause injuries elsewhere in their range may be expected to have similar effects if they are introduced to Australia.

(v) Role as disease carriers and reservoirs

Diseases spread from exotic fish to native fish may have huge ecological consequences (Hoffman and Schubert 1984, Shotts and Gratzek 1984, Taylor et al 1984, Langdon 1990, Arthington 1991, Lever 1996, Kailola 2000, Elvira 2001). Disease agents may include viruses, bacteria, protozoa, fungi and parasites. Little is known about diseases and parasites associated with aquarium fish (McKay 1984).

Risk assessment significance: It is difficult to predict the role that exotic species may have as vectors or reservoirs of diseases or parasites in new environments. However, species that harbour or transmit diseases/ parasites elsewhere may transmit the same agents if they are present in Australia.

(vi) Hybridisation with native species and other genetic changes

When exotic fish hybridise with native fish and produce fertile offspring, this hybridisation corrupts the gene pool of the native fish and hence may pose a threat to their survival (Taylor et al 1984; Arthington 1989, 1991; Crowl et al 1992; Lever 1996; Williamson 1996; Arthington and McKenzie 1997; Elvira 2001).

Fish are generally more plastic in their potential for hybridising than are mammals, and fewer crosses between fish species result in sterile progeny (Welcomme 1988). Hybrids may be produced spontaneously and survive in the wild. Through the removal of geographic barriers that normally prevent mixing of taxa, or under the pressures exerted through introductions that change normal behaviour patterns, hybrids arise between species or genera that would not otherwise interbreed (Elvira 2001). For example, the marbled trout (*Salmo marmoratus*) is endemic to rivers in the Adriatic Basin in Europe. Brown trout (*S. trutta*) were stocked there in 1906, leading to hybridisation between the two species and the near disappearance of marbled trout (Elvira 2001). Maciolek (1984) reported crosses between largemouth bass (*Micropterus salmoides*) and bluegill (*Lepomis macrochirus*) in at least two Hawaiian reservoirs.

According to Kailola (1989), if rainbowfish (Melanotaeniidae) were introduced to Australia from New Guinea, they might hybridise with Australian native species. Williamson (1996) reported that negative effects have been recorded in all known cases of hybridisation between introduced freshwater fish and native species.

Some ecologists have suggested fish taxa that freely hybridise in the wild (such as cichlids) may produce fertile hybrids that are a greater pest threat than the parent stock, because of hybrid vigour and enhanced reproductive potential (Kailola 2000). However, this theory is untested.

Exotic fish can have genetic effects other than hybridisation. Changes in the genetic structure of a population can occur due to reductions in its size, or reduced numbers of subpopulations or phenotypes caused by competition, habitat alterations or predation following the introduction of exotic species (Elvira 2001).

Risk assessment significance: Exotic species that have close relatives among Australia's endemic fish could hybridise with these native species and corrupt their gene pool.

4.5.4 Other factors having potential value for assessing the risk of impacts by introduced exotic fish

(i) History of being a pest overseas

Fish that are pests overseas may well become pests if they establish in Australia. Simple predictions can be made by assuming that invaders will cause significant impacts in a new area where they have established if they have already done so in other regions (Townsend and Winterbourn 1992, Ricciardi and Rasmussen 1998).

While correlative analyses are often limited by a scarce amount of comparable quantitative data, they can give an indication of potential impacts (Ricciardi and Rasmussen 1998). However, a species' history of impacts elsewhere is not an infallible guide to its potential impact in Australia. There are many examples in the scientific literature of species that have developed new behaviour and new dietary preferences when introduced to new environments and hence had impacts that could not have been predicted from their history. Hence, species that have little effect in their native (or previously introduced) range may have devastating effects when introduced to a new country (Bomford 2003, Hayes and Sliwa 2003). A further problem is that many potential pest species may not have been introduced outside their natural range yet, and so have not had the opportunity to demonstrate pest potential.

Risk assessment significance: Descriptive information on the impacts of previous invasions may provide a basis for useful predictions, although with a high degree of uncertainty. A precautionary approach is advisable for fish species that have no history of establishing outside their natural range.

(ii) Rate of spread

Species that spread rapidly from their initial place of establishment are likely to be harder to eradicate, contain or control, than species with a slow rate of spread. They are more likely to become widespread and to be considered pests. The factors that influence the rate of spread, and the final geographic range of an exotic species established in a new environment may differ from the factors that influence the probability of the initial establishment (Duncan et al 2001, Kolar and Lodge 2002, Forsyth et al 2004). Kolar and Lodge (2002) found exotic fish that spread rapidly in the Great Lakes of North America had slower growth rates, poorer survival in high water temperatures and greater temperature range tolerance than slowly spreading fish.

Risk assessment significance: There are inadequate data on rates of spread to enable this factor to be used to predict the pest potential of future fish introductions to Australia. However, fish that are known to have spread rapidly following their release into new environments overseas should be considered to pose a high risk.

(iii) Socio-economic effects

While significant recreational and commercial fisheries have developed from introduced fish such as trout, exotic fish species not favoured for human consumption can replace species that are popular for fishing (Welcomme 1988, Lever 1996, Elvira 2001). For example, the Mozambique tilapia established in reservoirs in India where they replaced more-favoured native species such as some carp (Welcomme 1988, Lever 1996). Nile perch introduced to Lake Victoria in Africa destroyed pre-existing sustainable fishing for a range of native species (Fryer 1991). According to Fryer (1991), it is doubtful if current levels of Nile perch fishing are sustainable or provide equivalent local benefits compared to pre-existing fisheries.

European carp introduced to Australia are claimed to cause problems by eroding banks of irrigation channels and blocking irrigation machinery (Koehn et al 2000). These carp are also said to have had detrimental impacts on some native species that are used for recreational and commercial fishing, but reliable data on these effects are unavailable (Koehn et al 2000).

Risk assessment significance: Introduced fish may bring economic benefits or cause economic harm. Because the distribution, abundance, sustainable harvest levels and impacts on other fish species of introduced fish are hard to predict accurately, forecasting the economic consequences of fish introductions to Australia is difficult. An examination of the economic consequences of previous introduction of a species elsewhere in the world may provide some indication of the potential consequences if a given species was introduced to Australia.

(iv) Similar appearance to harmful species

If a permitted species could be readily confused with undesirable or prohibited fish species at the size it is imported, this confusion could facilitate accidental importation of the harmful species (Kailola 1989). For example, small piranha (*Serrasalmus* spp) might be illegally imported in bags containing large numbers of the silver dollar (*Metynnis* sp) (McKay 1984).

Risk assessment significance: The risk of accidental entry of unwanted species through ports of entry will be determined by the adequacy of resources and expertise of quarantine authorities at these ports. In the future, it may be possible to undertake DNA testing of fish proposed for import at reasonable cost, since tests are now being developed for commercial use (Dr Nic Bax, personal communication 2004).

(v) Taxa

Kailola (2000) categorised the exotic fish families present in Australia taxa according to the level of risk they posed to native fish species and the environment. She considered the highest risk taxa were Poeciliids and Cyprinids, followed by Salmonids, Percids and Cichlids (moderate risk), and Cobitids and Belontiids (lowest risk). Kailola (2000) presented considerable anecdotal evidence on the impact of the fish taxa she assessed, although she considered there was insufficient information about the latter three taxa to fully assess risk. The review was restricted to exotic fish taxa already present in Australia. There are many other taxa with a record of having significant detrimental impacts on native species, including extinctions, where they are introduced. Examples include:

- round goby (family Gobiidae) — a piscivore (Ricciardi 2003)
- goby *Glossogobius giuris* (family Gobiidae) — a piscivore that also feeds on small insects and crustaceans (De Silva 1989)

- icefish *Neosalanx taihuensis* (family Salangidae) — a filter feeder that competes for food (Yang 1996)
- Nile perch (family Centropomidae) — a piscivore (Welcomme 1988).

Risk assessment significance: A detailed review of the literature on impacts of exotic fish worldwide might enable a ranking by taxa of risk of environmental (and economic and social) impacts. However, a species' history of impacts elsewhere is not an infallible guide to its potential impact in Australia. Such a review was beyond the scope of the current project.

(vi) Abundance

Elvira (2001) suggested that fish species associated with high impacts tend to have abundant populations in their native habitats.

Risk assessment significance: Few data are available on fish abundance, and fish in new habitats can reach densities much higher than those in their natural range. Therefore, this factor is not considered to be of value for predicting risk of impact.

(vi) Other factors

Kolar and Lodge (2002) found exotic fish that were considered to be a nuisance (pest) in the Great Lakes of North America had smaller eggs, wider salinity tolerances, and better survival in low water temperatures than non-nuisance fish. Kolar and Lodge (2002) found that these factors, which were correlated with nuisance status, differed from factors found to be correlated with establishment success.

Risk assessment significance: Further research is needed on these factors to see if they also apply to fish that are considered pests in other locations.

4.6 Discussion of factors affecting pest status for introduced freshwater fish

Unfortunately, relatively little research has been conducted on the impacts of exotic fish. Except for obvious species extinctions or economic losses, few studies have examined the possible suite of community changes that an invasive species can have (Cassey and Arthington 1999). European carp (*Cyprinus carpio*) and *Gambusia* spp (to a much lesser extent) are exceptions. As these species are the only two of the exotic finfish species established in Australia that can be assessed for impact, Kailola (2000) considered that neither meaningful categories nor comparisons can be made.

According to Elvira (2001), there are too few data to demonstrate how introduced species affect native species — thus, it is not possible to make rational decisions about which species are safe to import.

The impacts of exotic fish are most readily recognised when an abundant introduced species leads to major declines in native fish species, or causes obvious habitat alterations. Less obvious and less studied impacts include:

- competitive interactions that limit resource availability to native species
- changes to food web structures
- genetic alterations
- changes in abundance of lower-order taxa and lower trophic-level species.

Defining harmful species and identifying species that cause, or can potentially cause, ecological harm is inevitably a subjective process (Hayes and Sliwa 2003). Ecological harm is difficult to define and evaluate when it refers to species that are of no direct economic value, or to impacts on community structures and ecosystem processes. Such impacts are time consuming and hence expensive to evaluate, are often hampered by a lack of pre-invasion data, and therefore are largely under-reported in the scientific literature. Hence, some exotic species are perceived as having little obvious impact. There is no universally agreed formula to measure the environmental harm caused by introduced species and hence opinions on the type, extent and significance of impacts vary and even conflict (Hayes and Sliwa 2003).

Moyle and Light (1996b) suggested that most successful invasions by exotic fish do not have major community effects on recipient ecosystems, and that where such effects do occur, they are generally observed where species richness is low. However, these observations may simply be because in more diverse communities, impacts are less obvious, or may take longer to occur (Moyle and Light 1996b). Also, there are exceptions to this generalisation, such as in Lake Victoria in Africa, where the introduced piscivorous Nile perch eliminated over 200 species of endemic haplochromine cichlids (Welcomme 1988). Similarly, a goby, *Glossogobius giuris*, caused the extinction of 17 endemic cyprinid species in Lake Lanao in the Philippines (De Silva 1989: 146). Hence, Moyle and Light's (1996b) generalisation should not be used as grounds for assuming that most fish introductions in diverse communities will not have adverse ecological effects. As a general rule, it is best to assume we know too little about which communities are most vulnerable, and that interactions are too complex for community diversity to be a useful predictive factor for risk assessment.

Many exotic fish initially establish in highly disturbed and polluted habitats, often in or around urban areas. Such habitats are probably so degraded that they retain few native biota of conservation significance. However, exotic species that establish in these environments may act as sources for eventual spread to other habitats, where they have the potential to be a much higher threat to native species.

Since Australian aquatic systems are inherently different from overseas ones, there are limits to the usefulness of extrapolations drawn from overseas studies to Australian conditions (King 1995). Australian freshwater systems in particular differ markedly from those overseas. Not only is Australia one of the driest continents in the world, but Australian river flows are among the most variable. Australian waters also differ chemically from many other countries, with most water bodies being more saline and turbid than overseas examples. Biological differences are also significant, with peak litter fall in Australia occurring in summer instead of the northern hemisphere autumn, and this litter being mainly coarse woody material (King 1995). These differences could all affect the impacts of introduced fish.

Fish may show adaptive changes following colonisation events, to better suit them to their new environment (Arthington 1991). Shifts in thermal tolerance have been recorded for several species, including mosquitofish (*G. holbrooki*) and rainbow trout (*Oncorhynchus mykiss*) (Arthington 1991). Hybridisation between different strains of introduced species can lead to new genetic strains that are more invasive or have higher pest potential than the parent strains. An example is the Boolara strain of European carp (*C. carpio*) in Australia (Koehn et al 2000).

Moyle (1986), Moyle and Williams (1990) and Moyle and Light (1996a) suggested that native fish are most typically extirpated from waters that have been heavily modified by human activity, where native fish assemblages have already been depleted, disrupted or stressed. These authors recognised that exotic fish can establish in undisturbed areas, but considered that the native fish in such systems are usually able to adjust to the invader, and so extinctions following invasions of undisturbed systems are rare (Moyle and Light 1996a). However, they suggested that exceptions to this generalisation may occur when the introduced fish is a piscivore, or when it is capable of hybridising with the resident native species. This theory requires more study before it can be confirmed.

In summary, there is insufficient reliable knowledge of the factors correlated with impacts of exotic fish to make the development of a quantitative model feasible for assessing the risks of impacts of new exotic fish in Australia. Nonetheless, the review of factors associated with adverse impacts indicates that fish with the following attributes may be of greater risk of causing harm (with the caveat that fish with an absence of these factors cannot be taken to indicate that there is a low risk of harm):

- have adverse impacts elsewhere
- have close relatives with similar behavioural and ecological strategies that have adverse impacts elsewhere
- are generalist feeders
- are piscivorous
- destroy or modify aquatic vegetation
- stir up sediments to increase turbidity
- have potential to cause physical injury
- harbour or transmit diseases or parasites that are present in Australia
- have close relatives among Australia's endemic fish
- are known to have spread rapidly following their release into new environments.

This list could be used as a checklist to make a qualitative assessment of the threat of impacts posed by the establishment of new exotic fish species in Australia. Such an assessment would be particularly desirable if decisions are being made on whether to import species of exotic fish that score an Establishment Risk Rank of Moderate or higher in the quantitative risk assessment models presented in Sections 4.3 and 4.4.

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Appendix A: Scores for birds and mammals introduced to Australia

Table A1. Scores related to risk of establishment for exotic mammals and birds introduced to Australia

Introduced species	Climate Match PC Closest Standard Match Sum level 6	Overseas range size (million km²)	1. Climate Match Score (1–6)	2. Exotic Population Overseas Score (0–4)	3. Taxon Score (0–1)	4. Migration Score (0–1)	5. Diet Score (0–1)	6. Habitat Score (0–1)	7. Overseas range size Score (0–2)	Establishment Risk Score for Model 2 (0–16)	Establishment Risk Score for Model 1 (0–13)
Successful mammals											
Bos javanicus	607	1	3	2	1	1	1	1	0	9	6
Bos taurus	1088	2	4	4	1	1	1	1	1	13	10
Bubalus bubalis	931	4	4	4	1	1	1	1	1	13	10
Camelus dromedarius	154	3	2	0	1	1	1	1	1	7	4
Canis lupus	2780	197	6	4	1	1	1	1	2	16	13
Capra hircus	2250	10	5	4	1	1	1	1	1	14	11
Axis axis	1989	2	5	4	1	1	1	0	1	13	11
Cervus elaphus	1087	36	4	4	1	0	1	0	1	11	10
Axis porcinus	665	3	3	2	1	1	1	1	1	10	7
Cervus timorensis	119	1	2	4	1	1	1	1	0	10	7
Cervus unicolor	1035	5	4	4	1	1	1	1	1	13	10
Dama dama	731	11	3	4	1	0	1	1	1	11	9
Equus asinus	1841	8	5	4	1	1	1	1	1	14	11
Equus caballus	1177	9	4	4	1	1	1	1	1	13	10
Felis catus	2772	15	6	4	1	1	1	1	1	15	12
Funambulus pennanti	943	2	4	0	1	1	1	1	1	9	6
Lepus europeus	2718	7	6	4	1	1	1	1	1	15	12
Mus domesticus	2778	162	6	4	1	1	1	1	2	16	13
Oryctolagus cuniculus	696	8	3	4	1	1	1	1	1	12	9
Ovis aries	595	6	2	4	1	1	1	1	1	11	8

Introduced species	Climate Match PC Closest Standard Match Sum level 6	Overseas range size (million km²)	1. Climate Match Score (1–6)	2. Exotic Population Overseas Score (0–4)	3. Taxon Score (0–1)	4. Migration Score (0–1)	5. Diet Score (0–1)	6. Habitat Score (0–1)	7. Overseas range size Score (0–2)	Establishment Risk Score for Model 2 (0–16)	Establishment Risk Score for Model 1 (0–13)
Rattus norvegicus	2768	100	6	4	1	1	1	1	2	16	13
Rattus rattus	2776	58	6	4	1	1	1	1	1	15	12
Sus scrofa	2766	76	6	4	1	1	1	1	2	16	13
Vulpes vulpes	2591	175	5	4	1	1	1	1	2	15	12
Failed mammals											
Alces alces	101	69	2	0	1	0	1	0	1	5	4
Antilope cervicapra	1943	1	5	2	1	1	1	1	0	11	8
Canis aureus	2054	20	5	0	1	1	1	1	1	10	7
Capreolus capreolus	759	48	3	0	1	1	1	1	1	8	5
Cervus duvauceli	185	1	2	0	1	0	0	0	0	3	3
Cervus mariannus	21	1	1	0	1	0	0	1	0	3	2
Cervus nippon	256	6	2	4	1	0	1	1	1	10	8
Equus burchellii	1622	6	4	0	1	0	1	0	1	7	6
Herpestes edwardsi	1351	4	4	4	1	1	1	1	1	13	10
Herpestes javanicus	1328	6	4	2	1	1	1	1	1	11	8
Hydropotes inermis	42	1	1	4	1	1	1	1	0	9	6
Lama guanicoe	611	1	3	0	1	1	1	1	0	7	4
Vicugna vicugna	52	1	1	0	1	1	1	1	0	5	2
Mesocricetus auratus	482	1	2	0	1	1	1	0	0	5	3
Moschus moschiferus	69	22	1	0	1	1	1	0	1	5	3
Mustela putorius	684	15	3	2	1	1	1	1	1	10	7
Sciurus carolinensis	575	6	2	2	1	1	1	1	1	9	6
Suncus murinus	1074	14	4	4	1	1	1	1	1	13	10
Syncernus caffer	1523	8	4	0	1	1	1	1	1	9	6

Introduced species	Climate Match PC Closest Standard Match Sum level 6	Overseas range size (million km²)	1. Climate Match Score (1–6)	2. Exotic Population Overseas Score (0–4)	3. Taxon Score (0–1)	4. Migration Score (0–1)	5. Diet Score (0–1)	6. Habitat Score (0–1)	7. Overseas range size Score (0–2)	Establishment Risk Score for Model 2 (0–16)	Establishment Risk Score for Model 1 (0–13)
Taurotragus oryx	1140	5	4	0	1	0	1	0	1	7	6
Moschiola meminna	636	1	3	0	1	1	1	0	0	6	4
Successful birds											
Acridotheres tristis	1224	11	4	4	0	1	1	1	1	12	9
Alauda arvensis	787	57	3	4	0	0	1	1	1	10	8
Anas platyrhynchos	1902	95	5	4	0	0	1	1	2	13	11
Ardeola ibis	2746	54	6	4	0	0	1	1	1	13	11
Carduelis carduelis	1433	33	4	4	0	0	1	1	1	11	9
Carduelis chloris	765	21	3	4	0	0	1	1	1	10	8
Columba livia	2769	80	6	4	0	1	1	1	2	15	12
Cygnus olor	479	11	2	4	0	1	1	1	1	10	7
Lonchura punctulata	991	9	4	2	0	1	1	1	1	10	7
Passer domesticus	2772	110	6	4	0	1	1	1	2	15	12
Passer montanus	1434	70	4	4	0	1	1	1	2	13	10
Pavo cristatus	962	4	4	4	0	1	1	1	1	12	9
Pycnonotus jocosus	996	6	4	4	0	1	1	1	1	12	9
Streptopelia chinensis	1059	11	4	4	0	1	1	1	1	12	9
Streptopelia decaocto	2036	23	5	4	0	1	1	1	1	13	10
Streptopelia senegalensis	2717	44	6	4	0	1	1	1	1	14	11
Struthio camelus	1945	8	5	0	0	1	1	1	1	9	6
Sturnus vulgaris	2639	58	5	4	0	0	1	1	1	12	10
Turdus merula	1977	31	5	4	0	0	1	1	1	12	10
Turdus philomelos	644	31	3	4	0	0	1	1	1	10	8
Failed birds											
Acanthis cannabina	762	25	3	0	0	0	1	1	1	6	10
Agapornis roseicollis	1506	2	4	0	0	1	1	0	1	7	8

Introduced species	Climate Match PC Closest Standard Match Sum level 6	Overseas range size (million km²)	1. Climate Match Score (1–6)	2. Exotic Population Overseas Score (0–4)	3. Taxon Score (0–1)	4. Migration Score (0–1)	5. Diet Score (0–1)	6. Habitat Score (0–1)	7. Overseas range size Score (0–2)	Establishment Risk Score for Model 2 (0–16)	Establishment Risk Score for Model 1 (0–13)
Aix galericulata	75	3	1	4	0	0	1	1	1	8	10
Alectoris barbara	435	3	2	2	0	1	1	1	1	8	5
Alectoris Chukar	1592	23	4	4	0	1	1	0	1	11	9
Alectoris rufa	668	1	3	4	0	1	1	1	0	10	7
Branta canadensis	185	38	2	4	0	0	1	1	1	9	7
Callipepla californicus	1029	3	4	4	0	1	1	1	1	12	9
Carduelis spinus	542	22	2	0	0	0	1	1	1	5	3
Corvus splendens	1028	5	3	4	0	1	1	1	1	11	8
Emberiza citrinella	591	47	2	4	0	0	1	1	1	9	7
Emberiza hortulana	722	30	3	0	0	0	1	1	1	6	4
Erithacus rubecula	714	25	3	0	0	0	1	1	1	6	8
Euplectes albonotatus	2037	6	5	0	0	1	1	0	1	8	7
Euplectes orix	2601	13	5	0	0	1	1	0	1	8	4
Fringilla coelebs	762	27	3	4	0	0	1	1	1	10	8
Fringilla montifringilla	26	31	1	0	0	0	1	1	1	4	7
Gallus gallus	719	6	3	4	0	1	1	1	1	11	4
Lonchura malacca	978	7	4	4	0	1	1	1	1	12	9
Lophophorus impejanus	648	1	3	0	0	1	1	0	0	5	3
Lophura ignita	0	1	1	0	0	1	1	0	0	3	1
Lophura nycthemera	298	2	2	0	0	1	1	0	1	5	3
Luscinia megarhynchos	753	16	3	0	0	0	1	1	1	6	4
Numida meleagris	2415	20	5	4	0	1	1	1	1	13	10
Oena capensis	2631	20	5	0	0	1	1	1	1	9	6
Padda oryzivora	105	1	2	4	0	1	1	1	0	9	6

Introduced species	Climate Match PC Closest Standard Match Sum level 6	Overseas range size (million km²)	1. Climate Match Score (1–6)	2. Exotic Population Overseas Score (0–4)	3. Taxon Score (0–1)	4. Migration Score (0–1)	5. Diet Score (0–1)	6. Habitat Score (0–1)	7. Overseas range size Score (0–2)	Establishment Risk Score for Model 2 (0–16)	Establishment Risk Score for Model 1 (0–13)
Perdix perdix	689	32	3	4	0	1	1	1	1	11	8
Phasianus colchicus	2380	34	5	4	0	1	1	1	1	13	10
Plectropterus gambensis	2601	16	5	0	0	1	1	1	1	9	6
Pterocles exustus	1615	15	4	0	0	1	1	1	1	8	5
Pycnonotus cafer	948	6	4	4	0	1	1	1	1	12	9
Pyrrhula pyrrhula	614	41	3	0	0	1	1	1	1	7	4
Serinus canarius	48	0	1	2	0	1	1	1	0	6	3
Streptopelia turtur	1451	34	4	0	0	1	1	1	1	8	5

Scores were calculated using the instructions presented in Section 2.5.

Appendix B: Scores for birds and mammals introduced to New Zealand

Table B1. Scores related to risk of establishment for exotic mammals introduced to New Zealand

Introduced mammal	Scientific name[a]	Climate Match Score	Exotic Elsewhere Score	Overseas Range Size Score	Establishment Risk Score	Establishment Risk Rank
Successful mammals[a]						
Feral cattle	*Bos taurus*	1	4	0	5	Serious
Feral goat	*Capra hircus*	0	4	0	4	Serious
Red deer	*Cervus elaphus*	2	4	2	8	Extreme
Sika deer	*Cervus nippon*	2	4	0	6	Serious
Rusa deer	*Cervus timorensis*	1	4	0	5	Serious
Sambar deer	*Cervus unicolor*	0	4	0	4	Serious
Fallow deer	*Dama dama*	3	4	1	8	Extreme
Feral horse	*Equus caballus*	3	4	1	8	Extreme
Hedgehog	*Erinaceus europaeus*	4	3	2	9	Extreme
Feral cat	*Felis catus*	5	4	2	11	Extreme
Himalayan tahr	*Hemitragus jemlahicus*	0	4	0	4	Serious
European hare	*Lepus europaeus*	5	4	2	11	Extreme
Tammar (dama) wallaby	*Macropus eugenii*	1	2	0	3	Moderate
Red-necked (Bennet's) wallaby	*Macropus rufogriseus*	3	4	0	7	Extreme
House mouse	*Mus domesticus*	5	4	2	11	Extreme
Stoat, ermine	*Mustela erminea*	4	3	2	9	Extreme
Weasel	*Mustela nivalis*	3	0	2	5	Serious
Polecat, ferret	*Mustela putorius*	3	4	1	8	Extreme
White-tailed deer	*Odocoilus virginianus*	1	4	2	7	Extreme
European rabbit	*Oryctolagus cuniculus*	5	4	1	10	Extreme
Feral sheep	*Ovis aries*	1	4	0	5	Serious
Kiore	*Rattus exulans*	0	3	0	3	Moderate
Norway rat	*Rattus norvegicus*	4	4	2	10	Extreme

Introduced mammal	Scientific name[a]	Climate Match Score	Exotic Elsewhere Score	Overseas Range Size Score	Establishment Risk Score	Establishment Risk Rank
Black rat	*Rattus rattus*	5	4	2	11	Extreme
Chamois	*Rupicapra rupicapra*	1	4	0	5	Serious
Feral pig, wild boar	*Sus scrofa*	4	4	2	10	Extreme
Brush-tailed possum	*Trichosurus vulpecula*	4	2	0	6	Serious
Failed mammals[a]						
Moose	*Alces alces*	1	0	2	3	Moderate
Axis deer[b]	*Axis axis*[b]	1	4	0	5	Serious
Guinea pig	*Cavia porcellus*	1	3	0	4	Serious
Quoll[c]	*Dasyurus sp*[c]	-	2	0	-	-
Southern brown bandicoot	*Isoodon obesulus*	3	3	0	6	Serious
Alpaca	*Lama vicugna*	0	0	0	0	Low
Mule deer	*Odocoilus hemionus*	1	0	1	2	Moderate
Racoon	*Procyon lotor*	1	4	1	6	Serious
Common ringtail possum	*Pseudocheirus peregrinus*	3	3	0	6	Serious
Bharal (blue sheep)	*Pseudois nayaur*	0	2	0	2	Moderate
California ground squirrel	*Spermophilus beecheyi*	0	2	0	2	Moderate
Eastern chipmunk	*Tamais striatus*	1	0	0	1	Low

Scores were calculated according to the instructions presented in Section 2.6.
[a]Species sourced from Thomson (1922), Wodzicki (1950), Long (2003), King (2005) and Atkinson (2006).
[b]Excluded from Bomford et al's (unpublished data) analysis — established a small self-sustaining wild population that was eradicated by hunting.
[c]Excluded from Bomford et al's (unpublished data) analysis — unknown species. The quoll species with largest overseas range size (*Dasyurus geoffroii*) is used in this table.

Table B2. Scores related to risk of establishment for exotic birds introduced to New Zealand

Introduced bird	Scientific name[a]	Climate match score	Exotic Elsewhere Score	Migration score	Establishment Risk Score	Establishment Risk Rank
Successful birds[a]						
Redpoll	*Acanthis flammea*	0	2	2	4	Low
Common myna	*Acridotheres tristis*	2	4	2	8	High
Skylark	*Alauda arvensis*	2	4	2	8	High
Chukar partridge	*Alectoris graeca*	5	4	2	11	Extreme
Mallard	*Anas platyrhynchos*	1	4	2	7	Moderate
Cattle egret	*Ardeola ibis*	4	4	2	10	Extreme
Little owl	*Athene noctua*	4	4	2	10	Extreme
Canada goose	*Branta canadensis*	3	4	2	9	High
Sulphur-crested cockatoo	*Cacatua galerita*	4	2	2	8	High
Goldfinch	*Carduelis carduelis*	2	4	2	8	High
Greenfinch	*Carduelis chloris*	5	4	2	11	Extreme
Bobwhite quail	*Colinus virginianus*	5	4	2	11	Extreme
Rock dove	*Columba livia*	5	4	2	11	Extreme
Rook	*Corvus frugilegus*	3	2	2	7	Moderate
Brown quail	*Coturnix australis*	4	2	2	8	High
Black swan	*Cygnus atratus*	4	2	2	8	High
Mute swan	*Cygnus olor*	2	4	2	8	High
Kookaburra	*Dacelo novaeguineae*	4	4	2	10	Extreme
Yellowhammer	*Emberiza citrinella*	4	2	2	8	High
Cirl bunting	*Embiriza cirlus*	3	2	2	7	Moderate
Chaffinch	*Fringilla coelebs*	4	4	2	10	Extreme
Australian magpie	*Gymnorhyna tibicen*	4	2	2	8	High
California quail	*Lophortyx californicus*	1	4	2	7	Moderate
Turkey	*Meleagris gallopavo*	2	3	2	7	Moderate
House sparrow	*Passer domesticus*	5	4	2	11	Extreme
Peafowl	*Pavo cristatus*	2	4	2	8	High

Introduced bird	Scientific name[a]	Climate match score	Exotic Elsewhere Score	Migration score	Establishment Risk Score	Establishment Risk Rank
Ring-necked pheasant	*Phasianus colchicus*	3	4	2	9	High
Eastern rosella	*Platycercus eximius*	3	2	2	7	Moderate
Hedge sparrow	*Prunella modularis*	4	0	2	6	Moderate
Spotted turtledove	*Streptopelia chinensis*	3	4	2	9	High
Collared turtledove (Barbary dove)	*Streptopelia roseogrisea*	3	4	2	9	High
European starling	*Sturnus vulgaris*	5	4	2	11	Extreme
Blackbird	*Turdus merula*	5	4	2	11	Extreme
Song thrush	*Turdus philomelos*	5	4	2	11	Extreme
Silvereye	*Zosterops lateralis*	4	3	2	9	High
Failed birds[a]						
Linnet	*Acanthis cannabina*	4	0	2	6	Moderate
Twite	*Acanthis flavirostris*	2	2	2	6	Moderate
Red-browed waxbill	*Aegintha temporalis*	3	2	2	7	Moderate
Red-winged blackbird	*Agelaius phoeniceus*	1	2	2	5	Low
American wood duck	*Aix sponsa*	1	4	0	5	Low
Red-legged partridge	*Alectoris rufa*	3	4	2	9	High
Brush turkey	*Alectura lathami*	1	4	2	7	Moderate
Egyptian goose	*Alopochen aegyptianicus*	1	4	2	7	Moderate
Common pintail	*Anas acuta*	2	3	0	5	Low
European widgeon	*Anas Penelope*	2	2	0	4	Low
Greylag goose	*Anser anser*	1	2	0	3	Low
Snow goose	*Anser caerulescens*	0	2	0	2	Low
Common pochard	*Aythya ferina*	2	2	0	4	Low
Tufted duck	*Aythya fuligula*	2	0	0	2	Low
Hawaiian goose	*Branta sandvicensis*	0	2	2	4	Low
Siskin	*Carduelis spinus*	2	0	0	2	Low
Cape Barren goose	*Cereopsis novaehollandiae*	0	2	2	4	Low
Jackdaw	*Corvus monedula*	3	2	2	7	Moderate
Stubble quail	*Coturnix pectoralis*	3	0	0	3	Low

Introduced bird	Scientific name[a]	Climate match score	Exotic Elsewhere Score	Migration score	Establishment Risk Score	Establishment Risk Rank
Ortolan bunting	*Emberiza hortulanus*	3	0	0	3	Low
Reed bunting	*Emberiza shoeniclus*	4	2	2	8	Moderate
Diamond firetail	*Emblema guttata*	2	0	2	4	Low
European robin	*Erithacus rebecula*	4	0	2	6	Moderate
Bramble finch	*Fringilla montifringilla*	0	0	0	0	Low
Red jungle fowl	*Gallus gallus*	0	4	2	6	Moderate
Australian magpie lark	*Grallina cyanoleuca*	4	0	2	6	Moderate
Willow grouse	*Lagopus lagopus*	2	0	2	4	Low
Wonga pigeon	*Leucosarcia melanoleuca*	3	0	2	5	Low
Chestnut-breasted finch	*Lonchura castaneothorax*	1	3	2	6	Moderate
Nutmeg mannikin	*Lonchura punctulata*	0	4	2	6	Moderate
Silver pheasant	*Lophura nycthemera*	0	0	2	2	Low
Wood lark	*Lullula arborea*	3	0	2	5	Low
Nightingale	*Luscinia megarhynchos*	3	0	0	3	Low
Superb blue wren	*Malurus cyaneus*	4	2	2	8	Moderate
Noisy miner	*Manorina melanocephala*	3	0	2	5	Low
Bell miner	*Manorina melanophrys*	3	2	2	7	Moderate
Budgerigar	*Melopsittacus undulatus*	1	4	2	7	Moderate
Helmeted guineafowl	*Numida meleagris*	1	4	2	7	Moderate
Nankeen night heron	*Nycticorax caledonicus*	4	2	2	8	Moderate
Crested pigeon	*Ocyphaps lophotes*	1	2	2	5	Low
Mountain quail	*Oreortyx pictus*	1	3	2	6	Moderate
Java sparrow	*Padda oryzivora*	0	4	2	6	Moderate
Tree sparrow	*Passer montanus*	4	4	2	8	Moderate
Sharp-tailed grouse	*Pediocetes phasianellus*	0	0	2	2	Low
European partridge	*Perdix perdix*	3	4	2	9	High
Common bronzewing	*Phaps chalcoptera*	4	0	2	6	Moderate
Summer tanager	*Piranga rubra*	0	2	0	2	Low
Crimson rosella	*Platycercus elegans*	4	3	2	9	High

Introduced bird	Scientific name[a]	Climate match score	Exotic Elsewhere Score	Migration score	Establishment Risk Score	Establishment Risk Rank
Golden plover	*Pluvialis apricaria*	1	2	0	3	Low
Grey plover	*Pluvialis squatarola*	2	2	0	4	Low
Zebra finch	*Poephila guttata*	2	3	2	7	Moderate
Pintailed sandgrouse	*Pterocles alchata*	1	0	2	3	Low
Bullfinch	*Pyrrhula pyrrhula*	4	0	2	6	Moderate
Canary	*Serinus canarius*	0	3	2	5	Low
Javan turtledove	*Streptopelia bitorquata*	0	3	2	5	Low
Tawny owl	*Strix aluco*	3	2	2	7	Moderate
Western meadowlark	*Sturnella neglecta*	1	3	2	6	Moderate
Blackcap	*Sylvia atricapilla*	4	0	2	6	Moderate
Whitethroat	*Sylvia communis*	4	2	0	6	Moderate
Reeves pheasant	*Syrmaticus reevesii*	0	2	2	4	Low
Black grouse	*Tetrao tetrix*	2	0	2	4	Low
Greater prairie chicken	*Tympanuchus cupido*	0	0	2	2	Low
Barn owl	*Tyto alba*	5	3	2	10	Extreme
Lapwing	*Vanellus vanellus*	3	2	2	7	Moderate

Scores were calculated according to the instructions presented in Section 2.6.

[a]Species sourced from Veltman et al (1996) with an additional eight successful and ten failed species (from Long 1981 with outcome verified by C. J. R. Robertson, personal communication 2007).

Appendix C: Scores for reptiles and amphibians introduced to Australia, Britain, California and Florida

Table C1. Scores related to risk of establishment for exotic reptiles and amphibians (combined)[a] introduced to Britain, California and Florida calculated using the Australian Reptile and Amphibian Model (Bomford et al 2005, Bomford 2006)

Introduced species	Family	Climate matches Sum level 7	A. Climate Match Risk Score[b] (0–100)	B. Exotic Elsewhere Risk Score (0–30)	C. Taxonomic Family Risk Score (0–30)	Establishment Risk Score (0–160)	Establishment Risk Rank
Britain successful species							
Alytes obstetricians	Alytidae	190	98	30	15	143	Extreme
Zamenis longissima	Colubridae	161	83	0	10	93	Serious
Podarcis muralis	Lacertidae	190	98	30	20	148	Extreme
Rana lessonae	Ranidae	192	99	30	30	159	Extreme
Rana ridibunda	Ranidae	189	97	30	30	157	Extreme
Trituris alpestris	Salamandridae	190	98	30	15	143	Extreme
Triturus carnifex	Salamandridae	101	52	30	15	97	Serious
Xenopus laevis	Pipidae	94	48	30	15	93	Serious
Britain failed species							
Bombina bombina	Bombinatoridae	121	62	0	15	77	Serious
Bufo viridus	Bufonidae	134	69	30	20	119	Extreme
Chalcides ocellatus	Scincidae	5	3	30	15	48	Moderate
Chelydra serpentina	Chelydridae	0	0	30	10	40	Moderate
Chrysemys picta	Emydidae	0	0	15	15	30	Moderate
Coluber jugularis	Colubridae	3	02	0	10	12	Low
Discoglossus pictus	Discoglossidae	76	39	30	15	84	Serious
Eleutherodactylus johnstonei	Leptodactylidae	0	0	30	30	60	Moderate

Introduced species	Family	Climate matches Sum level 7	A. Climate Match Risk Score[b] (0–100)	B. Exotic Elsewhere Risk Score (0–30)	C. Taxonomic Family Risk Score (0–30)	Establishment Risk Score (0–160)	Establishment Risk Rank
Emys orbicularis	Emydidae	172	89	30	15	134	Extreme
Hydromantes genei	Plethodontidae	0	0	0	20	20	Low
Hyla aborea	Hylidae	192	99	0	15	114	Serious
Hyla meridionalis	Hylidae	64	33	30	15	114	Serious
Lacerta bilineata	Lacertidae	180	93	30	20	143	Extreme
Lampropeltis triangulum	Colubridae	8	04	0	10	14	Low
Litoria ewingii	Hylidae	185	95	30	15	140	Extreme
Natrix maura	Colubridae	129	66	30	10	106	Serious
Natrix tessellata	Colubridae	91	47	0	10	57	Serious
Pelobates fuscus	Pelobatidae	188	97	0	0	97	Serious
Pelodiscus sinensis	Trionychidae	0	0	30	20	50	Moderate
Podarcis dugesii	Lacertidae	0	0	30	20	50	Moderate
Podarcis sicula	Lacertidae	0	0	30	20	50	Moderate
Pseudacris regilla	Hylidae	45	23	30	15	68	Serious
Pseudocordylus microlepidotus	Cordylidae	0	0	0	10	10	Low
Rana pipiens	Ranidae	0	0	30	30	60	Moderate
Salamandra salamandra	Salamandridae	190	98	0	15	113	Serious
Scinax rubra	Hylidae	0	0	30	15	45	Moderate
Tarentola delalandii	Gekkonidae	0	0	0	30	30	Moderate
Tarentola mauritanica	Gekkonidae	0	0	30	30	60	Moderate
Terrapene carolina	Emydidae	0	0	30	15	45	Moderate
Testudo graeca	Testudinidae	3	2	30	15	47	Moderate
Thamnophis sirtalis	Colubridae	0	0	0	10	10	Low
Timon lepidus	Lacertidae	108	56	0	20	76	Serious

Introduced species	Family	Climate matches Sum level 7	A. Climate Match Risk Score[b] (0–100)	B. Exotic Elsewhere Risk Score (0–30)	C. Taxonomic Family Risk Score (0–30)	Establishment Risk Score (0–160)	Establishment Risk Rank
California successful species							
Ambystoma tigrinum	Ambystomatidae	24	14	30	15	59	Moderate
Apalone spinifera	Trionychidae	19	11	30	20	61	Serious
Chamaeleo jacksonii	Chamaeleontidae	0	0	30	30	60	Moderate
Chelydra serpentina	Chelydridae	1	1	30	10	41	Moderate
Hemidactylus turcicus	Gekkonidae	138	80	30	30	140	Extreme
Nerodia fasciata	Colubridae	0	0	30	10	40	Moderate
Rana berlandieri	Ranidae	10	6	30	30	66	Moderate
Rana catesbeiana	Ranidae	116	67	30	30	127	Extreme
Tarentola mauritanica	Gekkonidae	87	51	30	30	111	Serious
Trachemys scripta	Emydidae	22	13	30	15	58	Moderate
Xenopus laevis	Pipidae	72	42	30	15	87	Serious
California failed species							
Alligator mississipiensis	Alligatoridae	0	0	15	10	25	Moderate
Andrias japonicus	Cryptobranchidae	0	0	0	0	0	Low
Anolis carolinensis	Iguanidae	1	1	30	20	51	Moderate
Boa constrictor	Boidae	4	2	30	5	37	Moderate
Bufo marinus	Bufonidae	1	1	30	20	51	Moderate
Caiman crocodilus	Alligatoridae	0	0	30	15	45	Moderate
Mauremys reevesii	Geoemydidae	8	5	0	0	5	Low
Corallus hortulanus	Boidae	0	0	0	5	5	Low
Cordylus giganteus	Cordylidae	0	0	0	10	10	Low
Ctenosaura hemilopha	Iguanidae	0	0	30	20	50	Moderate
Drymarchon corais	Colubridae	0	0	0	10	10	Low
Elaphe guttata	Colubridae	1	1	30	10	41	Moderate
Eumeces obsoletus	Scincidae	1	1	0	15	16	Moderate

Introduced species	Family	Climate matches Sum level 7	A. Climate Match Risk Score[b] (0–100)	B. Exotic Elsewhere Risk Score (0–30)	C. Taxonomic Family Risk Score (0–30)	Establishment Risk Score (0–160)	Establishment Risk Rank
Gehyra mutilata	Gekkonidae	0	0	30	30	60	Moderate
Geochelone carbonaria	Testudinidae	0	0	0	15	15	Low
Graptemys pseudogeographica	Emydidae	0	0	30	15	45	Moderate
Heloderma horridum	Helodermatidae	0	0	0	0	0	Low
Hemidactylus garnotii	Gekkonidae	0	0	30	30	60	Moderate
Hemiphyllodactylus typus	Gekkonidae	0	0	30	30	60	Moderate
Hyla wrightorum	Hylidae	5	3	0	15	18	Low
Iguana iguana	Iguanidae	0	0	30	20	50	Moderate
Lampropeltis triangulum	Colubridae	4	2	0	10	12	Low
Lamprophis fuliginosus	Colubridae	51	3	0	10	40	Moderate
Lepidodactylus lugubris	Gekkonidae	60	35	30	30	95	Moderate
Leptodeira annulata	Colubridae	0	0	0	10	10	Low
Macrochelys temminckii	Chelydridae	0	0	0	10	10	Low
Malaclemys terrapin	Emydidae	0	0	15	15	30	Moderate
Naja haje	Elapidae	103	6	0	10	70	Serious
Nerodia sipedon	Colubridae	0	0	0	10	10	Low
Notophthalmus viridescens	Salamandridae	0	0	0	15	15	Low
Opheodrys aestivus	Colubridae	0	0	0	10	10	Low
Palea steindachneri	Trionychidae	0	0	30	20	50	Moderate
Phrynosoma cornutum	Iguanidae	1	1	30	20	51	Moderate
Pseudemys concinna	Emydidae	0	0	30	15	45	Moderate
Pseudemys floridana	Emydidae	0	0	0	15	15	Low
Python molurus	Pythonidae	0	0	30	5	35	Moderate
Python reticulatus	Pythonidae	0	0	0	5	5	Low
Sceloporus jarrovii	Iguanidae	1	1	0	20	21	Low
Sceloporus poinsettii	Iguanidae	1	1	0	20	21	Low

Introduced species	Family	Climate matches Sum level 7	A. Climate Match Risk Score[b] (0–100)	B. Exotic Elsewhere Risk Score (0–30)	C. Taxonomic Family Risk Score (0–30)	Establishment Risk Score (0–160)	Establishment Risk Rank
Sceloporus serrifer	Iguanidae	0	0	0	20	20	Low
Terrapene carolina	Emydidae	0	0	30	15	45	Moderate
Terrapene ornata	Emydidae	10	6	0	15	21	Low
Thamnophis sauritus	Colubridae	0	0	15	10	25	Moderate
Varanus salvator	Varanidae	0	0	0	15	15	Low
Florida successful species							
Agama agama	Agamidae	11	1	30	30	61	Serious
Ameiva ameiva	Teiidae	54	51	0	20	71	Serious
Anolis chlorocyanus	Iguanidae	8	8	30	20	58	Moderate
Anolis cristatellus	Iguanidae	7	7	30	20	57	Moderate
Anolis cybotes	Iguanidae	8	8	30	20	58	Moderate
Anolis distichus	Iguanidae	29	27	30	20	77	Serious
Anolis equestris	Iguanidae	38	36	30	20	86	Serious
Anolis ferreus	Iguanidae	3	3	0	20	23	Moderate
Anolis garmani	Iguanidae	51	48	15	20	83	Serious
Anolis porcatus	Iguanidae	45	42	30	20	92	Serious
Anolis sagrei	Iguanidae	104	98	30	20	148	Extreme
Aspidoscelis motaguae	Teiidae	0	0	0	20	20	Low
Basiliscus vittatus	Iguanidae	20	19	0	20	39	Moderate
Boa constrictor	Boidae	100	94	30	5	129	Extreme
Caiman crocodilus	Alligatoridae	8	8	30	10	48	Moderate
Calotes versicolor	Agamidae	70	66	30	30	126	Extreme
Chamaeleo calyptratus	Chamaeleontidae	0	0	30	30	60	Moderate
Cnemidophorus lemniscatus	Teiidae	4	4	30	20	54	Moderate
Cosymbotus platyurus	Gekkonidae	78	74	0	30	104	Serious
Ctenosaura pectinata	Iguanidae	0	0	30	20	50	Moderate

Introduced species	Family	Climate matches Sum level 7	A. Climate Match Risk Score[b] (0–100)	B. Exotic Elsewhere Risk Score (0–30)	C. Taxonomic Family Risk Score (0–30)	Establishment Risk Score (0–160)	Establishment Risk Rank
Ctenosaura similis	Iguanidae	7	7	0	20	27	Moderate
Eleutherodactylus coqui	Leptodactylidae	1	1	30	30	61	Serious
Eleutherodactylus planirostris	Leptodactylidae	76	72	30	30	132	Extreme
Gekko gecko	Gekkonidae	77	73	15	30	118	Serious
Gonatodes albogularis	Gekkonidae	46	43	30	30	103	Serious
Hemidactylus frenatus	Gekkonidae	76	72	30	30	132	Extreme
Hemidactylus garnotii	Gekkonidae	89	84	30	30	144	Extreme
Hemidactylus mabouia	Gekkonidae	37	35	30	30	95	Serious
Hemidactylus turcicus	Gekkonidae	78	74	30	30	134	Extreme
Iguana iguana	Iguanidae	80	75	30	20	125	Extreme
Leiocephalus carinatus	Iguanidae	57	54	0	20	74	Serious
Leiocephalus schreibersi	Iguanidae	5	5	0	20	25	Moderate
Mabuya multifasciata	Scincidae	89	84	30	15	129	Extreme
Osteopilus septentrionalis	Hylidae	45	42	30	15	86	Serious
Chondrodactylus bibronii	Gekkonidae	4	4	0	30	34	Moderate
Phelsuma madagascariensis	Gekkonidae	7	7	30	30	67	Serious
Python molurus	Pythonidae	68	64	0	5	69	Serious
Ramphotyphlops braminus	Typhlopidae	89	84	30	30	144	Extreme
Sphaerodactylus elegans	Gekkonidae	45	42	0	30	72	Serious
Tarentola annularis	Gekkonidae	4	4	0	30	34	Moderate
Tarentola mauritanica	Gekkonidae	0	0	30	30	60	Moderate
Florida failed species							
Anolis conspersus	Iguanidae	6	6	0	20	26	Moderate
Atelopus zeteki	Bufonidae	0	0	0	20	20	Low
Basiliscus basiliscus	Iguanidae	0	0	0	20	20	Low
Bufo arenarum	Bufonidae	80	75	0	20	95	Serious
Bufo blombergi	Bufonidae	0	0	0	20	20	Low

Introduced species	Family	Climate matches Sum level 7	A. Climate Match Risk Score[b] (0–100)	B. Exotic Elsewhere Risk Score (0–30)	C. Taxonomic Family Risk Score (0–30)	Establishment Risk Score (0–160)	Establishment Risk Rank
Bufo schneideri	Bufonidae	97	92	0	20	112	Serious
Chelus fimbriatus	Chelidae	16	15	0	10	25	Moderate
Cordylus cordylus	Cordylidae	0	0	0	10	10	Low
Cyclura cornuta	Iguanidae	8	8	0	20	28	Moderate
Cynops pyrrhogaster	Salamandridae	0	0	0	15	15	Low
Hemidactylus brookii	Gekkonidae	74	7	30	30	67	Serious
Hymenochirus boettgeri	Pipidae	3	3	0	15	18	Low
Kinosternon scorpioides	Kinosternidae	25	24	0	0	24	Moderate
Pachymedusa dacnicolor	Hylidae	0	0	0	15	15	Low
Podocnemis lewyana	Pelomedusidae	0	0	0	10	10	Low
Podocnemis sextuberculata	Pelomedusidae	0	0	0	10	10	Low
Podocnemis unifilis	Pelomedusidae	29	27	0	10	37	Moderate
Python regius	Pythonidae	0	0	0	5	5	Low
Python reticulatus	Pythonidae	9	8	0	5	13	Low
Sphaerodactylus macrolepis	Gekkonidae	3	3	0	30	33	Moderate
Trachemys dorbigni	Emydidae	1	1	0	15	16	Low
Trachemys stejnegeri	Emydidae	7	7	30	15	52	Moderate
Tupinambis nigropunctatus	Teiidae	58	55	0	20	75	Serious
Typhlops lumbricalis	Typhlopidae	45	42	0	30	72	Serious
Varanus exanthematicus	Varanidae	5	5	0	15	20	Low
Varanus salvator	Varanidae	44	42	0	15	57	Moderate
Xenopus laevis	Pipidae	26	25	30	15	70	Serious

Scores were calculated using the instructions presented in Section 3.4.

[a]Sources: Fred Kraus unpublished database of literature records; Kevin M. Enge (Florida Fish and Wildlife Conservation Commission, personal communication 15 March 2005) list of exotic species established in Florida for at least ten years; Meshaka et al (2004). Reproduced from Bomford et al (2005).

[b]The Climate Match Scores have not been corrected for small numbers of input meteorological stations where this is applicable. For Florida the successful species that had 12 or fewer meteorological stations in their overseas geographic range were: *Anolis chlorocyanus, A. ferreus, A. garmani* and *Leiocephalus schreibersi* and the failed species were: *Anolis conspersus, Atelopus zetiki, Bufo blombergi, Podocnemis lewyana* and *P. sextuberculata*. To correct for bias introduced due to few input meteorological stations, all these nine species should have ten points added to both their Climate Match Scores and their Establishment Risk Scores.

Table C2. Scores related to risk of establishment for exotic reptiles and amphibians (combined) introduced to the Australian mainland calculated using the Australian Reptile and Amphibian Model model (Section 3.4, Bomford et al 2005, Bomford 2006)

Introduced species	Family	Climate match PC Euclidian Sum level 7	A: Climate Match Risk Score (0–100)	B: Exotic Elsewhere Risk Score (0–30)	C: Taxonomic Family Risk Score (0–30)	Establishment Risk Score (0–160)	Establishment Risk Rank
Successful species							
Cane toad (*Bufo marinus*)	Bufonidae	1849	66	30	20	116	Extreme
Asian house gecko (*Hemidactylus frenatus*)	Gekkonidae	698	25	30	30	85	Serious
Mourning gecko (*Lepidodactylus lugubris*)	Gekkonidae	79	3	30	30	63	Serious
Red-eared slider (*Trachemys scripta*)	Emydidae	1504	54	30	15	99	Serious
Flowerpot snake (*Ramphotyphlops braminus*)	Typhlopidae	947	34	30	30	94	Serious
Failed species							
Axolotl or salamander (*Ambystoma mexicanum*)	Ambystomatidae	0	0	0	15	15	Low
Black-spined toad (*Bufo melanostictus*)	Bufonidae	967	35	30	20	85	Serious

Scores were calculated using the instructions presented in Section 3.4.

Table C3. Scores related to probability of establishment for reptiles and amphibians introduced to Britain, California and Florida[a] calculated using the Reptile and Amphibian Model (Bomford et al 2008)

Introduced species	Family	World range size[b]	Climate 6 score	*Prop. species*	*Prop.genus*	*Prop.family*	Probability of establishment
Britain failed species[c]							
Bombina bombina	Bombinatoridae	7.0	0.887	0.250	0.222	0.222	0.258
Bufo viridus	Bufonidae	12.5	0.912	0.750	0.620	0.602	0.363
Chalcides ocellatus	Scincidae	16.5	0.330	0.500	0.667	0.535	0.013
Chelydra serpentina	Chelydridae	9.4	0.201	0.333	0.333	0.250	0.011
Chrysemys picta	Emydidae	10.9	0.010	0.100	0.100	0.364	0.001
Discoglossus pictus	Alytidae	0.3	0.778	1.000	0.667	0.571	0.652
Elaphe obsoleta	Colubridae	4.3	0.010	0.200	0.219	0.199	0.005
Eleutherodactylus johnstonei	Leptodactylidae	<0.001	0.000	0.923	0.807	0.333	0.061
Emys orbicularis	Emydidae	9.2	0.985	0.467	0.467	0.364	0.438
Hemidactylus turcicus	Gekkonidae	17.1	0.196	0.900	0.881	0.757	0.013
Hyla arborea	Hylidae	6.1	0.995	0.000	0.429	0.388	0.201
Hyla meridionalis	Hylidae	0.8	0.758	0.571	0.429	0.388	0.377
Lampropeltis getula	Colubridae	6.5	0.031	0.000	0.000	0.199	0.002
Lampropeltis triangulum	Colubridae	9.7	0.407	0.000	0.000	0.199	0.010
Litoria ewingii	Hylidae	0.6	1.000	0.667	0.303	0.388	0.709
Mauremys caspica	Geoemydidae	3.8	0.196	0.000	0.150	0.364	0.004
Natrix maura	Colubridae	1.5	0.959	0.400	0.200	0.199	0.560
Natrix tessellata	Colubridae	13.0	0.830	0.200	0.200	0.199	0.124
Pelobates fuscus	Pelobatidae	13.2	0.995	0.000	0.000	0.000	0.201
Pelodiscus sinensis	Trionychidae	2.8	0.000	0.714	0.714	0.525	0.015
Podarcis dugesii	Lacertidae	0.07	0.000	0.667	0.692	0.565	0.014
Podarcis sicula	Lacertidae	0.3	0.206	0.800	0.692	0.565	0.057
Pseudacris regilla	Hylidae	2.5	0.701	0.571	0.364	0.388	0.269
Rana pipiens	Ranidae	14.3	0.031	0.500	0.742	0.748	0.002
Salamandra salamandra	Salamandridae	7.6	0.995	0.000	0.000	0.306	0.150
Scinax rubra	Hylidae	4.3	0.139	0.750	0.667	0.388	0.040

Introduced species	Family	World range size[b]	Climate 6 score	*Prop. species*	*Prop.genus*	*Prop.family*	Probability of establishment
Tarentola delalandii	Gekkonidae	0.03	0.000	0.000	0.680	0.757	0.001
Tarentola mauritanica	Gekkonidae	1.1	0.216	0.750	0.680	0.757	0.029
Terrapene carolina	Emydidae	1.5	0.031	0.067	0.050	0.364	0.002
Testudo graeca	Testudinidae	3.1	0.206	0.400	0.382	0.323	0.020
Thamnophis sirtalis	Colubridae	16.0	0.010	0.000	0.059	0.199	0.001
Timon lepidus	Lacertidae	1.4	0.959	0.000	0.214	0.565	0.133
Britain successful species[c]							
Alytes obstetricians	Alytidae	2.5	0.995	0.333	0.333	0.429	0.418
Lacerta bilineata	Lacertidae	3.5	0.995	0.333	0.143	0.548	0.281
Podarcis muralis	Lacertidae	4.0	0.995	0.800	0.667	0.548	0.713
Rana esculenta	Ranidae	9.8	0.995	0.333	0.735	0.742	0.203
Rana lessonae	Ranidae	9.2	0.995	1.000	0.735	0.742	0.645
Rana perezi	Ranidae	1.1	0.789	1.000	0.735	0.742	0.556
Rana ridibunda	Ranidae	7.8	0.995	0.889	0.735	0.742	0.598
Trituris alpestris	Salamandridae	3.7	0.995	0.667	0.500	0.278	0.752
Triturus carnifex	Salamandridae	0.2	0.835	0.714	0.500	0.278	0.674
Xenopus laevis	Pipidae	12.5	0.871	0.444	0.474	0.455	0.197
Zamenis longissima	Colubridae	2.4	0.995	0.000	0.000	0.199	0.208
California failed species[d]							
Alligator mississipiensis	Alligatoridae	1.3	0.000	0.000	0.000	0.136	0.053
Anolis carolinensis	Iguanidae	0.8	0.151	0.706	0.691	0.549	0.395
Boa constrictor	Boidae	17.3	0.326	0.158	0.158	0.063	0.149
Bufo marinus	Bufonidae	16.2	0.337	0.704	0.620	0.602	0.270
Caiman crocodilus	Alligatoridae	10.4	0.012	0.273	0.273	0.136	0.075
Corallus caninus	Boidae	4.6	0.000	0.000	0.000	0.063	0.050
Drymarchon corais	Colubridae	1.0	0.058	0.000	0.000	0.199	0.062
Elaphe guttata	Colubridae	5.7	0.122	0.125	0.219	0.199	0.104
Gehyra mutilata	Gekkonidae	0.03	0.064	0.765	0.591	0.757	0.213

Introduced species	Family	World range size[b]	Climate 6 score	*Prop. species*	*Prop.genus*	*Prop.family*	Probability of establishment
Geochelone carbonaria	Testudinidae	0.2	0.000	0.000	0.500	0.323	0.059
Glyptemys muhlenbergii	Emydidae	0.3	0.000	0.000	0.000	0.364	0.032
Graptemys geographica	Emydidae	0.47	0.000	0.500	0.143	0.364	0.138
Graptemys pseudogeographica	Emydidae	4.2	0.000	0.111	0.143	0.364	0.037
Hemidactylus garnotii	Gekkonidae	2.9	0.064	0.778	0.881	0.757	0.231
Hemiphyllodactylus typus	Gekkonidae	0.01	0.064	0.857	0.857	0.757	0.319
Hyla wrightorum	Hylidae	0.2	0.058	0.000	0.429	0.388	0.062
Iguana iguana	Iguanidae	14.0	0.250	0.688	0.647	0.549	0.245
Kinosternon subrubrum	Kinosternidae	3.6	0.000	0.000	0.000	0.000	0.063
Lampropeltis triangulum	Colubridae	9.7	0.116	0.000	0.000	0.199	0.041
Lepidodactylus lugubris	Gekkonidae	1.4	0.512	0.842	0.680	0.757	0.764
Leptodeira annulata	Colubridae	14.1	0.227	0.000	0.000	0.199	0.049
Macrochelys temminckii	Chelydridae	2.4	0.000	0.000	0.000	0.250	0.036
Malaclemys terrapin	Emydidae	0.4	0.012	0.000	0.000	0.364	0.034
Mauremys reevesii	Geoemydidae	4.0	0.047	0.182	0.182	0.364	0.059
Notophthalmus viridescens	Salamandridae	10.5	0.081	0.000	0.000	0.306	0.024
Palea steindachneri	Trionychidae	4.5	0.064	0.667	0.667	0.525	0.222
Phrynosoma cornutum	Iguanidae	2.8	0.099	0.217	0.172	0.549	0.163
Pseudemys concinna	Emydidae	1.9	0.070	0.286	0.143	0.364	0.098
Pseudemys floridana	Emydidae	0.4	0.000	0.000	0.143	0.364	0.037
Python molurus	Pythonidae	4.6	0.012	0.143	0.036	0.063	0.080
Python reticulatus	Pythonidae	2.4	0.000	0.000	0.036	0.063	0.061
Sceloporus poinsettii	Iguanidae	0.9	0.093	0.000	0.250	0.549	0.092
Terrapene carolina	Emydidae	1.5	0.035	0.067	0.050	0.364	0.044
Terrapene ornata	Emydidae	2.2	0.058	0.000	0.050	0.364	0.038
Thamnophis sauritus	Colubridae	3.1	0.035	0.000	0.059	0.199	0.049
Varanus exanthematicus	Varanidae	6.7	0.169	0.000	0.467	0.467	0.053
Varanus salvator	Varanidae	5.2	0.000	0.000	0.467	0.467	0.026

Introduced species	Family	World range size[b]	Climate 6 score	*Prop. species*	*Prop.genus*	*Prop.family*	Probability of establishment
California successful species[d]							
Ambystoma tigrinum	Ambystomatidae	15.1	0.413	0.375	0.250	0.267	0.281
Apalone spinifera	Trionychidae	6.8	0.378	0.538	0.500	0.500	0.420
Chamaeleo jacksonii	Chamaeleontidae	0.8	0.163	1.000	0.769	0.647	0.573
Chelydra serpentina	Chelydridae	9.4	0.349	0.267	0.267	0.200	0.288
Eleutherodactylus coqui	Leptodactylidae	<0.001	0.000	0.667	0.789	0.000	0.586
Hemidactylus turcicus	Gekkonidae	17.1	0.930	0.867	0.873	0.753	0.907
Nerodia fasciata	Colubridae	2.1	0.000	1.000	0.167	0.193	0.489
Rana berlandieri	Ranidae	0.9	0.076	1.000	0.735	0.742	0.390
Rana catesbeiana	Ranidae	10.7	0.971	0.867	0.735	0.742	0.948
Rana sphenocephala	Ranidae	3.76	0.000	0.333	0.735	0.742	0.048
Tarentola mauritanica	Gekkonidae	1.1	0.773	0.700	0.640	0.753	0.890
Trachemys scripta	Emydidae	4.5	0.532	0.652	0.641	0.360	0.819
Xenopus laevis	Pipidae	12.5	0.715	0.444	0.444	0.455	0.671
Florida failed species[e]							
Cyclura cornuta	Iguanidae	0.003	0.406	0.000	0.667	0.549	0.337
Cynops pyrrhogaster	Salamandridae	<0.001	0.208	0.000	0.000	0.306	0.151
Geochelone denticulata	Testudinidae	7.3	0.726	1.000	0.500	0.323	0.975
Hemidactylus brookii	Gekkonidae	9.1	0.783	0.900	0.881	0.757	0.939
Laudakia stellio	Agamidae	2.2	0.000	0.800	0.800	0.655	0.316
Lissemys punctata	Trionychidae	1.78	0.057	0.250	0.250	0.525	0.095
Malayemys subtrijuga	Geoemydidae	0.68	0.387	0.000	0.000	0.364	0.264
Podocnemis unifilis	Pelomedusidae	5.2	0.708	0.000	0.250	0.333	0.639
Python regius	Pythonidae	3.0	0.038	0.000	0.036	0.063	0.105
Python reticulatus	Pythonidae	2.4	0.387	0.000	0.036	0.063	0.421
Trachemys stejnegeri	Emydidae	0.009	0.330	0.667	0.650	0.364	0.793
Tupinambis teguixin	Teiidae	8.4	0.962	0.500	0.333	0.800	0.866
Varanus exanthematicus	Varanidae	6.7	0.660	0.000	0.467	0.467	0.518
Varanus salvator	Varanidae	4.5	0.708	0.000	0.467	0.467	0.622
Xenopus laevis	Pipidae	12.5	0.783	0.500	0.500	0.500	0.831

Introduced species	Family	World range size[b]	Climate 6 score	*Prop. species*	*Prop.genus*	*Prop.family*	Probability of establishment
Florida successful species[e]							
Agama agama	Agamidae	9.2	0.660	0.333	0.250	0.621	0.545
Anolis chlorocyanus	Iguanidae	0.001	0.406	1.000	0.679	0.543	0.903
Anolis cristatellus	Iguanidae	<0.001	0.340	1.000	0.679	0.543	0.869
Anolis cybotes	Iguanidae	0.001	0.406	1.000	0.679	0.543	0.903
Anolis distichus	Iguanidae	<0.001	0.575	0.500	0.679	0.543	0.840
Anolis equestris	Iguanidae	<0.001	0.623	0.500	0.679	0.543	0.870
Anolis garmani	Iguanidae	<0.001	0.755	0.000	0.679	0.543	0.756
Anolis porcatus	Iguanidae	0.003	0.632	1.000	0.679	0.543	0.967
Anolis sagrei	Iguanidae	0.009	1.000	0.588	0.679	0.543	0.983
Boa constrictor	Boidae	17.3	0.991	0.105	0.105	0.047	0.873
Bufo marinus	Bufonidae	16.3	1.000	0.685	0.609	0.591	0.942
Caiman crocodilus	Alligatoridae	10.4	0.425	0.182	0.182	0.091	0.451
Calotes versicolor	Agamidae	3.9	0.802	0.875	0.889	0.621	0.973
Chamaeleo calyptratus	Chamaeleontidae	0.1	0.000	1.000	0.714	0.647	0.478
Chondrodactylus bibronii	Gekkonidae	0.9	0.519	0.500	0.333	0.753	0.589
Cnemidophorus lemniscatus	Teiidae	7.4	0.274	1.000	1.000	0.700	0.701
Cosymbotus platyurus	Gekkonidae	2.1	0.972	0.000	0.000	0.753	0.685
Ctenosaura pectinata	Iguanidae	0.003	0.000	1.000	0.600	0.543	0.520
Ctenosaura similis	Iguanidae	0.1	0.472	0.000	0.600	0.543	0.401
Eleutherodactylus coqui	Leptodactylidae	<0.001	0.113	0.667	0.789	0.000	0.797
Eleutherodactylus planirostris	Leptodactylidae	0.06	0.972	0.900	0.789	0.000	0.998
Gekko gecko	Gekkonidae	4.1	0.972	0.250	0.333	0.753	0.843
Gonatodes albogularis	Gekkonidae	0.4	0.632	0.833	0.889	0.753	0.926
Hemidactylus frenatus	Gekkonidae	5.7	0.972	0.880	0.873	0.753	0.981
Hemidactylus garnotii	Gekkonidae	2.9	0.972	0.667	0.873	0.753	0.972
Hemidactylus mabouia	Gekkonidae	13.4	0.953	1.000	0.873	0.753	0.972
Hemidactylus turcicus	Gekkonidae	17.0	0.991	0.867	0.873	0.753	0.954

Introduced species	Family	World range size[b]	Climate 6 score	*Prop. species*	*Prop.genus*	*Prop.family*	Probability of establishment
Iguana iguana	Iguanidae	14.0	0.981	0.625	0.588	0.543	0.944
Mabuya multifasciata	Scincidae	5.1	0.972	0.500	0.438	0.521	0.955
Osteopilus septentrionalis	Hylidae	<0.001	0.632	0.625	0.625	0.378	0.937
Phelsuma madagascariensis	Gekkonidae	0.6	0.302	1.000	0.867	0.753	0.782
Python molurus	Pythonidae	4.6	0.802	0.000	0.000	0.047	0.834
Ramphotyphlops braminus	Typhlopidae	7.5	0.972	0.971	0.971	0.928	0.975
Sphaerodactylus argus	Gekkonidae	<0.001	0.764	1.000	0.643	0.753	0.969
Tarentola mauritanica	Gekkonidae	1.21	0.000	0.700	0.640	0.753	0.196
Varanus niloticus	Varanidae	11.4	0.792	0.000	0.400	0.400	0.614

Values were calculated according to the instructions presented in Section 3.6.

[a]Sources: Fred Kraus unpublished database of literature records; Kevin M. Enge (Florida Fish and Wildlife Conservation Commission, personal communication 15 March 2005) list of exotic species established in Florida for at least ten years; Meshaka et al (2004). Reproduced from Bomford et al (2005).

[b]World geographic range size excluding jurisdiction being assessed (million km^2).

[c]For Britain (Scotland, England and Wales), only species introduced from outside the British Isles were included.

[d]For California, species translocated from the eastern states of the United States were included to increase the sample size, but any species originally native within California were excluded.

[e]For Florida, only exotic species introduced from outside the United States were included.

Table C4. Scores related to risk of establishment in Australia for exotic reptiles and amphibians previously introduced to Britain, California and Florida calculated using the Reptile and Amphibian Model (Bomford et al 2008)

Introduced Species	Family	Raw Sum Climate 6 Score for Australia	*Prop.species*	*S(Climate 6)* for Australia	*Family Random Effect*	Establishment Risk Score for Australia	Establishment Risk Rank for Australia
Rana berlandieri	Ranidae	1697	1.00	0.710	1.69	0.99	Extreme
Rana catesbeiana	Ranidae	2427	0.87	1.824	1.69	0.99	Extreme
Hemidactylus mabouia	Gekkonidae	2646	1.00	2.158	-0.41	0.98	Extreme
Anolis carolinensis	Iguanidae	2452	0.67	1.862	0.49	0.97	Extreme
Hemidactylus turcicus	Gekkonidae	2752	0.87	2.320	-0.41	0.97	Extreme
Rana perezi	Ranidae	1042	1.00	-0.290	1.69	0.97	Extreme
Ramphotyphlops braminus	Typhlopidae	2023	0.97	1.207	0.02	0.96	Extreme
Chelydra serpentina	Chelydridae	2664	0.31	2.185	0.68	0.95	Extreme
Rana lessonae	Ranidae	737	1.00	-0.755	1.69	0.95	Extreme
Xenopus laevis	Pipidae	2735	0.47	2.294	0.03	0.95	Extreme
Rana sphenocephala	Ranidae	1640	0.50	0.623	1.69	0.95	Extreme
Ambystoma tigrinum	Ambystomatidae	2293	0.44	1.619	0.64	0.94	Extreme
Anolis sagrei	Iguanidae	2086	0.61	1.303	0.49	0.94	Extreme
Apalone spinifera	Trionychidae	2550	0.57	2.011	-0.07	0.94	Extreme
Basiliscus vittatus	Iguanidae	1315	1.00	0.127	0.49	0.94	Extreme
Rana ridibunda	Ranidae	739	0.90	-0.752	1.69	0.94	Extreme
Trachemys scripta	Emydidae	2726	0.65	2.280	-0.77	0.93	Extreme
Iguana iguana	Iguanidae	1615	0.65	0.585	0.49	0.90	Extreme
Chondrodactylus bibronii	Gekkonidae	2121	0.67	1.357	-0.41	0.89	Extreme
Chalcides ocellatus	Scincidae	2472	0.33	1.892	-0.00	0.89	Extreme
Bufo marinus	Bufonidae	2370	0.69	1.737	-0.91	0.88	Extreme
Varanus niloticus	Varanidae	2697	0.33	2.236	-0.59	0.86	Extreme
Phrynosoma cornutum	Iguanidae	2190	0.21	1.462	0.49	0.85	Serious
Chamaeleo jacksonii	Chamaeleontidae	638	1.00	-0.906	0.48	0.84	Serious

Introduced Species	Family	Raw Sum Climate 6 Score for Australia	*Prop.species*	*S(Climate 6)* for Australia	*Family Random Effect*	Establishment Risk Score for Australia	Establishment Risk Rank for Australia
Hemidactylus frenatus	Gekkonidae	1458	0.88	0.345	-0.41	0.84	Serious
Rana pipiens	Ranidae	832	0.46	-0.610	1.69	0.83	Serious
Eleutherodactylus planirostris	Leptodactylidae	357	0.91	-1.335	1.08	0.83	Serious
Rana esculenta	Ranidae	738	0.50	-0.754	1.69	0.83	Serious
Podarcis muralis	Lacertidae	939	0.82	-0.447	0.34	0.81	Serious
Elaphe guttata	Colubridae	2551	0.12	2.013	-0.15	0.80	Serious
Tarentola mauritanica	Gekkonidae	1595	0.71	0.554	-0.41	0.80	Serious
Alytes obstetricians	Alytidae	1687	0.50	0.694	0.01	0.79	Serious
Ctenosaura pectinata	Iguanidae	395	1.00	-1.277	0.49	0.79	Serious
Calotes versicolor	Agamidae	957	0.89	-0.420	-0.11	0.78	Serious
Chamaeleo calyptratus	Chamaeleontidae	352	1.00	-1.343	0.48	0.78	Serious
Cosymbotus platyurus	Gekkonidae	953	1.00	-0.426	-0.41	0.78	Serious
Boa constrictor	Boidae	2336	0.15	1.685	-0.09	0.77	Serious
Nerodia fasciata	Colubridae	746	1.00	-0.742	-0.15	0.77	Serious
Anolis cristatellus	Iguanidae	262	1.00	-1.480	0.49	0.75	Serious
Eleutherodactylus johnstonei	Leptodactylidae	93	0.89	-1.738	1.08	0.75	Serious
Phelsuma madagascariensis	Gekkonidae	840	1.00	-0.598	-0.41	0.75	Serious
Lampropeltis triangulum	Colubridae	2567	0.00	2.037	-0.15	0.75	Serious
Hemidactylus brookii	Gekkonidae	1160	0.82	-0.110	-0.41	0.74	Serious
Pseudemys concinna	Emydidae	2464	0.25	1.880	-0.77	0.74	Serious
Lampropeltis getula	Colubridae	2481	0.00	1.906	-0.15	0.72	Serious
Podarcis sicula	Lacertidae	721	0.75	-0.780	0.34	0.72	Serious
Anolis porcatus	Iguanidae	121	1.00	-1.695	0.49	0.71	Serious
Triturus carnifex	Salamandridae	670	0.75	-0.858	0.35	0.70	Serious
Trituris alpestris	Salamandridae	663	0.75	-0.868	0.35	0.70	Serious
Sceloporus serrifer	Iguanidae	1985	0.00	1.149	0.49	0.70	Serious
Anolis cybotes	Iguanidae	53	1.00	-1.799	0.49	0.69	Serious

Introduced Species	**Family**	**Raw Sum Climate 6 Score for Australia**	***Prop.species***	***S(Climate 6)* for Australia**	***Family Random Effect***	**Establishment Risk Score for Australia**	**Establishment Risk Rank for Australia**
Sceloporus poinsettii	Iguanidae	1952	0.00	1.099	0.49	0.69	Serious
Anolis chlorocyanus	Iguanidae	25	1.00	-1.842	0.49	0.68	Serious
Chrysemys picta	Emydidae	2567	0.09	2.037	-0.77	0.67	Serious
Mabuya multifasciata	Scincidae	957	0.67	-0.420	-0.00	0.67	Serious
Notophthalmus viridescens	Salamandridae	1985	0.00	1.149	0.35	0.67	Serious
Hemidactylus garnotii	Gekkonidae	1119	0.70	-0.172	-0.41	0.66	Serious
Laudakia stellio	Agamidae	997	0.67	-0.359	-0.11	0.66	Serious
Cnemidophorus lemniscatus	Teiidae	706	1.00	-0.803	-0.77	0.63	Serious
Lacerta bilineata	Lacertidae	939	0.50	-0.447	0.34	0.63	Serious
Eleutherodactylus coqui	Leptodactylidae	27	0.70	-1.839	1.08	0.62	Serious
Discoglossus pictus	Alytidae	750	0.67	-0.735	0.01	0.60	Serious
Bufo viridus	Bufonidae	1484	0.60	0.385	-0.91	0.60	Serious
Elaphe obsolete	Colubridae	1791	0.17	0.853	-0.15	0.60	Serious
Pelodiscus sinensis	Trionychidae	775	0.67	-0.697	-0.07	0.59	Serious
Drymarchon corais	Colubridae	2098	0.00	1.322	-0.15	0.59	Serious
Emys orbicularis	Emydidae	1567	0.44	0.511	-0.77	0.55	Serious
Thamnophis sirtalis	Colubridae	1984	0.00	1.148	-0.15	0.55	Serious
Natrix tessellate	Colubridae	1646	0.17	0.632	-0.15	0.54	Serious
Anolis equestris	Iguanidae	200	0.67	-1.575	0.49	0.51	Serious
Ctenosaura similis	Iguanidae	488	0.50	-1.135	0.49	0.50	Serious
Agama agama	Agamidae	812	0.50	-0.641	-0.11	0.47	Serious
Sphaerodactylus argus	Gekkonidae	46	1.00	-1.810	-0.41	0.47	Serious
Lepidodactylus lugubris	Gekkonidae	420	0.80	-1.239	-0.41	0.47	Serious
Leptodeira annulata	Colubridae	1775	0.00	0.829	-0.15	0.47	Serious
Anolis distichus	Iguanidae	59	0.67	-1.790	0.49	0.46	Serious
Gonatodes albogularis	Gekkonidae	295	0.86	-1.430	-0.41	0.46	Serious
Alligator mississipiensis	Alligatoridae	1335	0.00	0.157	0.43	0.45	Serious

Introduced Species	Family	Raw Sum Climate 6 Score for Australia	*Prop.species*	*S(Climate 6)* for Australia	*Family Random Effect*	Establishment Risk Score for Australia	Establishment Risk Rank for Australia
Zamenis longissima	Colubridae	773	0.50	-0.700	-0.15	0.45	Serious
Kinosternon subrubrum	Kinosternidae	1612	0.00	0.580	-0.05	0.43	Serious
Natrix maura	Colubridae	1051	0.33	-0.276	-0.15	0.43	Serious
Bufo melanostictus	Bufonidae	1121	0.50	-0.169	-0.91	0.39	Moderate
Pseudacris regilla	Hylidae	1029	0.50	-0.310	-0.82	0.38	Moderate
Hemiphyllodactylus typus	Gekkonidae	256	0.75	-1.489	-0.41	0.37	Moderate
Gekko gecko	Gekkonidae	1034	0.33	-0.302	-0.41	0.36	Moderate
Lissemys punctata	Trionychidae	1045	0.20	-0.285	-0.07	0.36	Moderate
Terrapene carolina	Emydidae	1750	0.06	0.791	-0.77	0.35	Moderate
Caiman crocodilus	Alligatoridae	573	0.25	-1.006	0.43	0.34	Moderate
Gehyra mutilata	Gekkonidae	221	0.72	-1.543	-0.41	0.34	Moderate
Podarcis dugesii	Lacertidae	134	0.50	-1.676	0.34	0.34	Moderate
Terrapene ornate	Emydidae	1834	0.00	0.919	-0.77	0.34	Moderate
Anolis garmani	Iguanidae	22	0.50	-1.846	0.49	0.33	Moderate
Hyla meridionalis	Hylidae	880	0.50	-0.537	-0.82	0.33	Moderate
Python molurus	Pythonidae	1110	0.13	-0.186	-0.08	0.33	Moderate
Salamandra salamandra	Salamandridae	1067	0.00	-0.252	0.35	0.33	Moderate
Testudo graeca	Testudinidae	1436	0.38	0.311	-1.30	0.33	Moderate
Timon lepidus	Lacertidae	1052	0.00	-0.275	0.34	0.32	Moderate
Litoria ewingii	Hylidae	862	0.50	-0.565	-0.82	0.32	Moderate
Palea steindachneri	Trionychidae	206	0.50	-1.566	-0.07	0.27	Moderate
Hyla wrightorum	Hylidae	1630	0.00	0.607	-0.82	0.27	Moderate
Macrochelys temminckii	Chelydridae	636	0.00	-0.909	0.68	0.26	Moderate
Osteopilus septentrionalis	Hylidae	210	0.65	-1.560	-0.82	0.22	Moderate
Varanus exanthematicus	Varanidae	1321	0.00	0.136	-0.59	0.22	Moderate
Bombina bombina	Bombinatoridae	596	0.20	-0.970	-0.26	0.19	Moderate
Python reticulatus	Pythonidae	821	0.00	-0.627	-0.08	0.18	Moderate

Introduced Species	Family	Raw Sum Climate 6 Score for Australia	*Prop.species*	*S(Climate 6)* for Australia	*Family Random Effect*	Establishment Risk Score for Australia	Establishment Risk Rank for Australia
Tupinambis teguixin	Teiidae	601	0.33	-0.963	-0.77	0.17	Moderate
Scinax rubra	Hylidae	63	0.60	-1.784	-0.82	0.16	Low
Trachemys stejnegeri	Emydidae	31	0.57	-1.833	-0.77	0.15	Low
Geochelone denticulata	Testudinidae	434	0.50	-1.218	-1.30	0.13	Low
Thamnophis sauritus	Colubridae	633	0.00	-0.914	-0.15	0.13	Low
Pelobates fuscus	Pelobatidae	636	0.00	-0.909	-0.22	0.13	Low
Mauremys reevesii	Geoemydidae	758	0.00	-0.723	-0.37	0.13	Low
Ambystoma mexicanum	Ambystomatidae	0	0.00	-1.880	0.64	0.12	Low
Mauremys caspica	Geoemydidae	672	0.00	-0.855	-0.37	0.12	Low
Varanus salvator	Varanidae	845	0.00	-0.591	-0.59	0.12	Low
Cyclura cornuta	Iguanidae	36	0.00	-1.825	0.49	0.11	Low
Python regius	Pythonidae	393	0.00	-1.280	-0.08	0.10	Low
Hyla arborea	Hylidae	873	0.00	-0.548	-0.82	0.10	Low
Corallus caninus	Boidae	276	0.00	-1.459	-0.09	0.09	Low
Cynops pyrrhogaster	Salamandridae	11	0.00	-1.863	0.35	0.09	Low
Graptemys geographica	Emydidae	86	0.33	-1.749	-0.77	0.09	Low
Malayemys subtrijuga	Geoemydidae	477	0.00	-1.152	-0.37	0.09	Low
Leptophis depressirostris	Colubridae	0	0.00	-1.880	-0.15	0.06	Low
Podocnemis unifilis	Pelomedusidae	309	0.00	-1.408	-0.62	0.06	Low
Geochelone carbonaria	Testudinidae	693	0.00	-0.822	-1.30	0.05	Low
Graptemys pseudogeographica	Emydidae	187	0.10	-1.595	-0.77	0.05	Low
Tarentola delalandii	Gekkonidae	10	0.00	-1.865	-0.41	0.04	Low
Glyptemys muhlenbergii	Emydidae	96	0.00	-1.734	-0.77	0.04	Low
Malaclemys terrapin	Emydidae	189	0.00	-1.592	-0.77	0.04	Low
Pseudemys floridana	Emydidae	214	0.00	-1.553	-0.77	0.04	Low

Values were calculated using the instructions presented in Section 3.6. Species are listed in order of Establishment Risk Score.

Appendix D: Freshwater fish scores

Table D1. Scores related to risk of establishment for exotic freshwater fish introduced to Australia calculated using Exotic Freshwater Fish Model 1

Successfully introduced species	A	B	C	D	E	F	G
	Climate Match Score 1–8	Overseas Range Score 0–4	Establishment Score 0–3	Introduction Success Score 0–4	Taxa Risk Score 0–5	Total Establishment Risk Score[a] 0–24	Establishment Risk Rank
Three-spot cichlid *Cichlasoma trimaculatum*	4	2	0	0	4	10	Moderate
Victoria Burton's haplochromine *Haplochromis burtoni*	4	0	1	2	3	10	Moderate
Goby *Acentrogobius pflaumii*	3	0	1	2	5	11	Moderate
Blue acara *Aequidens pulcher*	4	2	2	2	3	13	Moderate
Red devil/Midas cichlid *Amphilophus citrinellus*	3	2	2	4	3	14	Moderate
Convict cichlid *iatus Archocentrus nigrofasciatus*	4	0	2	4	4	14	Moderate
Niger cichlid *Tilapia mariae*	2	3	2	4	4	15	Serious
White-cloud mountain minnow *Tanichthys albonubes*	3	4	2	4	3	16	Serious
Yellowfin goby *Acanthogobius flavimanus*	4	1	2	4	5	16	Serious
Chameleon goby *Tridentiger trigonocephalus*	5	1	2	4	5	17	Serious
One-spot live bearer *Phalloceros caudimaculatus*	4	3	2	4	5	18	Serious
Jack Dempsey *Cichlasoma octofasciatum*	5	2	3	4	4	18	Serious
Weather loach *Misgurnus anguillicaudatus*	4	2	3	4	5	18	Serious
Brook trout *Salvelinus fontinalis*	4	4	3	3	4	18	Serious
Roach *Rutilus rutilus*	5	4	3	4	3	19	Serious
Jewel cichlid *Hemichromis bimaculatus*	5	3	2	4	5	19	Serious
Sailfin molly *Poecilia latipinna*	6	2	3	4	5	20	Extreme
Platy *Xiphophorus maculatus*	6	2	3	4	5	20	Extreme
Green swordtail *Xiphophorus hellerii*	7	1	3	4	5	20	Extreme

Successfully introduced species	**A**	**B**	**C**	**D**	**E**	**F**	**G**
	Climate Match Score 1–8	**Overseas Range Score 0–4**	**Establishment Score 0–3**	**Introduction Success Score 0–4**	**Taxa Risk Score 0–5**	**Total Establishment Risk Score[a] 0–24**	**Establishment Risk Rank**
Redbelly tilapia *Tilapia zillii*	6	4	3	3	4	20	Extreme
Redfin perch *Perca fluviatilis*	5	3	3	4	5	20	Extreme
Tench *Tinca tinca*	6	3	3	4	5	21	Extreme
Oscar *Astronotus ocellatus*	5	4	3	4	5	21	Extreme
Rainbow trout *Oncorhynchus mykiss*	7	4	3	4	3	21	Extreme
Mosquitofish *Gambusia holbrooki + affinis*	6	4	3	4	5	22	Extreme
Guppy *Poecilia reticulata*	6	4	3	4	5	22	Extreme
Goldfish *Carassius auratus*	7	4	3	4	5	23	Extreme
Mozambique tilapia *Oreochromis mossambicus*	8	4	3	4	4	23	Extreme
European carp *Cyprinus carpio*	8	4	3	4	5	24	Extreme
Unsuccessfully introduced species (recorded but not known to be established)							
Sobaity seabream *Sparidentex hasta*	1	0	1	2	2	6	Low
Pearl cichlid[b] *Geophagus brasiliensis*	3	1	0	0	3	7	Low
Redbanded perch *Hypoplectrodes huntii*	2	0	1	2	2	7	Low
Japanese seabass *Lateolabrax japonicus*	3	2	1	2	0	8	Moderate
Common triplefin *Forsterygion lapillum*	4	0	1	2	2	9	Moderate
Dominican gambusia *Gambusia dominicensis*	2	0	1	2	5	10	Moderate
Green terror *Aequidens rivulatus*	3	1	1	2	3	10	Moderate
Banded cichlid *Heros severus*	4	2	0	0	4	10	Moderate
American flagfish *Jordanella floridae*	2	0	1	2	5	10	Moderate
Sumatra barb *Puntius tetrazona*	1	0	3	3	5	12	Moderate
Plainfin frogfish *Porichthys notatus*	4	4	1	2	2	13	Moderate
Redhead *Vieja synspila*	3	4	1	2	4	14	Moderate
Chinook salmon *Oncorhynchus tshawytscha*	4	4	2	1	3	14	Moderate
Firemouth cichlid *Thorichtys meeki*	5	0	3	4	4	16	Serious

Successfully introduced species	A	B	C	D	E	F	G
	Climate Match Score 1–8	**Overseas Range Score 0–4**	**Establishment Score 0–3**	**Introduction Success Score 0–4**	**Taxa Risk Score 0–5**	**Total Establishment Risk Score[a] 0–24**	**Establishment Risk Rank**
Atlantic salmon *Salmo salar*	5	4	3	1	4	17	Serious
Wami tilapia *Oreochromis urolepis*	7	1	3	4	4	19	Serious
Blue tilapia *Oreochromis aureus*	6	3	3	4	4	20	Extreme
Rosy barb [b] *Puntius conchonius*	6	3	3	4	5	21	Extreme

[a]The Total Establishment Risk Score (column F) is the sum of the scores in columns A–E.
[b]The pearl cichlid and rosy barb have since established exotic populations in Australia.

Table D2. Scores related to probability of establishment for exotic freshwater fish introduced to ten countries calculated using Exotic Freshwater Fish Model 2

Introduced species	Family	Outcome[a]	Climate 6	*Prop.species*	*Prop.genus*	*Prop.family*	Probability of establishment
Australia							
Aequidens rivulatus	Cichlidae	0	0.004	1.000	1.000	0.825	0.749
Gambusia dominicensis	Poeciliidae	0	0.009	1.000	0.966	0.931	0.812
Heros severus	Cichlidae	0	0.039	0.000	0.000	0.825	0.158
Oncorhynchus tshawytscha	Salmonidae	0	0.251	0.267	0.520	0.527	0.350
Oreochromis aureus	Cichlidae	0	0.341	0.880	0.859	0.825	0.913
Oreochromis urolepis	Cichlidae	0	0.062	1.000	0.859	0.825	0.818
Puntius tetrazona	Cyprinidae	0	0	0.500	0.846	0.728	0.329
Salmo salar	Salmonidae	0	0.167	0.357	0.620	0.527	0.335
Thorichtys meeki	Cichlidae	0	0.01	1.000	1.000	0.825	0.757
Acanthogobius flavimanus	Gobiidae	1	0.023	1.000	1.000	1.000	0.858
Aequidens pulcher	Cichlidae	1	0.014	0.000	0.000	0.822	0.133
Amphilophus citrinellus	Cichlidae	1	0.014	1.000	1.000	0.822	0.760
Archocentrus nigrofasciatus	Cichlidae	1	0.214	0.500	0.500	0.822	0.706
Astronotus ocellatus	Cichlidae	1	0.022	0.625	0.625	0.825	0.522
Carassius auratus	Cyprinidae	1	0.246	0.978	0.962	0.726	0.893
Cichlasoma octofasciatum	Cichlidae	1	0.073	1.000	0.650	0.822	0.823
Cichlasoma trimaculatum	Cichlidae	1	0.07	0.000	0.650	0.822	0.205
Cyprinus carpio	Cyprinidae	1	0.603	0.907	0.907	0.726	0.913
Gambusia holbrooki	Poeciliidae	1	0.339	1.000	0.966	0.924	0.954
Geophagus brasiliensis	Cichlidae	1	0.038	0.000	0.500	0.822	0.165
Hemichromis bimaculatus	Cichlidae	1	0.103	1.000	1.000	0.822	0.858
Misgurnus anguillicaudatus	Cobitidae	1	0.08	1.000	1.000	1.000	0.902
Oncorhynchus mykiss	Salmonidae	1	0.432	0.732	0.510	0.523	0.725
Oreochromis mossambicus	Cichlidae	1	0.942	0.940	0.853	0.822	0.960

Introduced species	Family	Outcome[a]	Climate 6	*Prop.species*	*Prop.genus*	*Prop.family*	Probability of establishment
Perca fluviatilis	Percidae	1	0.208	1.000	1.000	0.960	0.943
Phalloceros caudimaculatus	Poeciliidae	1	0.08	1.000	1.000	0.924	0.878
Poecilia latipinna	Poeciliidae	1	0.286	0.889	0.902	0.924	0.931
Poecilia reticulata	Poeciliidae	1	0.435	0.967	0.902	0.924	0.953
Puntius conchonius	Cyprinidae	1	0.282	0.750	0.833	0.725	0.824
Rutilus rutilus	Cyprinidae	1	0.22	0.750	0.667	0.726	0.791
Salmo trutta	Salmonidae	1	0.229	0.750	0.600	0.523	0.668
Salvelinus fontinalis	Salmonidae	1	0.197	0.639	0.589	0.523	0.562
Tanichthys albonubes	Cyprinidae	1	0.004	1.000	1.000	0.726	0.684
Tilapia mariae	Cichlidae	1	0.038	1.000	0.857	0.822	0.788
Tilapia zillii	Cichlidae	1	0.562	0.882	0.857	0.822	0.927
Tinca tinca	Cyprinidae	1	0.251	0.833	0.833	0.726	0.846
Trichogaster trichopterus	Belontiidae	1	0.245	0.714	0.800	0.806	0.831
Tridentiger trigonocephalus	Gobiidae	1	0.134	1.000	1.000	1.000	0.929
Xiphophorus hellerii	Poeciliidae	1	0.069	0.929	0.885	0.924	0.842
Xiphophorus maculatus	Poeciliidae	1	0.036	0.875	0.885	0.924	0.781
Italy							
Acipenser transmontanus	Acipenseridae	0	0.267	0.167	0.059	0.050	0.077
Anguilla japonica	Anguillidae	0	0.089	0.200	0.333	0.333	0.092
Aristichthys nobilis	Cyprinidae	0	0.069	0.310	0.310	0.728	0.303
Astronotus ocellatus	Cichlidae	0	0	0.750	0.750	0.825	0.575
Coregonus clupeaformis	Salmonidae	0	0.238	0.000	0.514	0.527	0.193
Oncorhynchus kisutch	Salmonidae	0	0.257	0.333	0.520	0.527	0.400
Oncorhynchus tshawytscha	Salmonidae	0	0.267	0.267	0.520	0.527	0.361
Osphronemus goramy	Osphronemidae	0	0	0.364	0.364	0.364	0.086
Salmo salar	Salmonidae	0	0.772	0.357	0.620	0.527	0.564
Salvelinus alpinus	Salmonidae	0	0.02	0.429	0.607	0.527	0.188
Ameiurus melas	Ictaluridae	1	0.287	1.000	0.948	0.838	0.935

Introduced species	Family	Outcome[a]	Climate 6	*Prop.species*	*Prop.genus*	*Prop.family*	Probability of establishment
Ameiurus natalis	Ictaluridae	1	0.277	1.000	0.948	0.838	0.933
Ameiurus nebulosus	Ictaluridae	1	0.525	0.946	0.948	0.838	0.940
Carassius auratus	Cyprinidae	1	0.931	0.978	0.962	0.726	0.951
Coregonus lavaretus	Salmonidae	1	0.218	0.714	0.486	0.523	0.631
Cyprinus carpio	Cyprinidae	1	0.95	0.907	0.907	0.726	0.942
Gambusia affinis	Poeciliidae	1	0.921	0.957	0.966	0.924	0.972
Lepomis auritus	Centrarchidae	1	0.297	1.000	0.857	0.767	0.919
Lepomis gibbosus	Centrarchidae	1	0.891	1.000	0.857	0.767	0.956
Micropterus salmoides	Centrarchidae	1	0.931	0.829	0.695	0.767	0.933
Odontesthes bonariensis	Atherinidae	1	0.208	1.000	1.000	1.000	0.950
Pachychilon pictum	Cyprinidae	1	0.257	1.000	1.000	0.726	0.903
Pseudorasbora parva	Cyprinidae	1	0.832	1.000	1.000	0.726	0.947
Rhodeus sericeus	Cyprinidae	1	0.515	1.000	1.000	0.726	0.927
Salvelinus fontinalis	Salmonidae	1	0.337	0.639	0.589	0.523	0.652
Sander lucioperca	Percidae	1	0.653	1.000	1.000	0.960	0.969
Silurus glanis	Siluridae	1	0.822	1.000	1.000	1.000	0.977
Ctenopharyngodon idella	Cyprinidae	1	0.287	0.537	0.537	0.726	0.711
Japan							
Catla catla	Cyprinidae	0	0	0.556	0.556	0.728	0.355
Coregonus clupeaformis	Salmonidae	0	0.188	0.000	0.514	0.527	0.164
Labeo rohita	Cyprinidae	0	0.034	0.250	0.333	0.728	0.221
Lepomis cyanellus	Centrarchidae	0	0.505	0.714	0.878	0.775	0.864
Morone saxatilis	Moronidae	0	0.438	0.400	0.400	0.385	0.390
Oncorhynchus mykiss	Salmonidae	0	0.356	0.750	0.520	0.527	0.726
Oncorhynchus nerka	Salmonidae	0	0.038	0.200	0.520	0.527	0.120
Oncorhynchus tshawytscha	Salmonidae	0	0.101	0.267	0.520	0.527	0.205
Oreochromis macrochir	Cichlidae	0	0	0.364	0.859	0.825	0.312
Osphronemus goramy	Osphronemidae	0	0	0.364	0.364	0.364	0.086
Pomoxis nigromaculatus	Centrarchidae	0	0.534	0.750	0.714	0.775	0.876
Sarotherodon galilaeus	Cichlidae	0	0	0.667	0.667	0.825	0.513

Introduced species	Family	Outcome[a]	Climate 6	*Prop.species*	*Prop.genus*	*Prop.family*	Probability of establishment
Sarotherodon melanotheron	Cichlidae	0	0	1.000	0.667	0.825	0.733
Aristichthys nobilis	Cyprinidae	1	0.399	0.286	0.286	0.726	0.571
Coregonus lavaretus	Salmonidae	1	0.063	0.714	0.486	0.523	0.412
Gambusia affinis	Poeciliidae	1	0.505	0.957	0.966	0.924	0.954
Lepomis macrochirus	Centrarchidae	1	0.663	0.846	0.857	0.767	0.914
Megalobrama amblycephala	Cyprinidae	1	0.288	0.000	0.000	0.726	0.327
Micropterus salmoides	Centrarchidae	1	0.572	0.829	0.695	0.767	0.899
Odontesthes bonariensis	Atherinidae	1	0.005	1.000	1.000	1.000	0.841
Oncorhynchus kisutch	Salmonidae	1	0.096	0.222	0.510	0.523	0.178
Oreochromis aureus	Cichlidae	1	0.005	0.840	0.853	0.822	0.646
Oreochromis mossambicus	Cichlidae	1	0.005	0.940	0.853	0.822	0.709
Oreochromis niloticus	Cichlidae	1	0.005	0.857	0.853	0.822	0.658
Rhodeus ocellatus	Cyprinidae	1	0.38	1.000	1.000	0.726	0.919
Salmo trutta	Salmonidae	1	0.293	0.750	0.600	0.523	0.707
Salvelinus fontinalis	Salmonidae	1	0.293	0.639	0.589	0.523	0.636
Salvelinus namaycush	Salmonidae	1	0.144	0.600	0.589	0.523	0.467
Tilapia sparrmanii	Cichlidae	1	0.005	1.000	0.857	0.822	0.743
Tilapia zillii	Cichlidae	1	0	0.882	0.857	0.822	0.666
Tinca tinca	Cyprinidae	1	0.274	0.833	0.833	0.726	0.854
Ctenopharyngodon idella	Cyprinidae	1	0.404	0.537	0.537	0.726	0.741
Hypophthalmichthys molitrix	Cyprinidae	1	0.274	0.674	0.674	0.726	0.784
Mylopharyngodon piceus	Cyprinidae	1	0.332	0.333	0.333	0.726	0.590
Morocco							
Aristichthys nobilis	Cyprinidae	0	0	0.310	0.310	0.728	0.207
Esox niger	Esocidae	0	0	1.000	0.818	0.818	0.733
Lates niloticus	Centropomidae	0	0	0.750	0.600	0.600	0.390
Oncorhynchus clarki	Salmonidae	0	0.068	0.00	0.520	0.527	0.087
Pomoxis annularis	Centrarchidae	0	0.023	1.00	0.714	0.775	0.736

Introduced species	Family	Outcome[a]	Climate 6	*Prop.species*	*Prop.genus*	*Prop.family*	Probability of establishment
Pomoxis nigromaculatus	Centrarchidae	0	0.545	0.75	0.714	0.775	0.877
Salmo trutta	Salmonidae	0	0.841	0.781	0.620	0.527	0.827
Salvelinus alpinus	Salmonidae	0	0	0.429	0.607	0.527	0.167
Salvelinus fontinalis	Salmonidae	0	0.136	0.667	0.607	0.527	0.507
Thymallus thymallus	Salmonidae	0	0.318	0.000	0.000	0.527	0.212
Barbus barbus	Cyprinidae	1	0	1.000	0.600	0.726	0.663
Cyprinus carpio carpio	Cyprinidae	1	0.818	0.907	0.907	0.726	0.930
Esox lucius	Esocidae	1	0.091	0.875	0.727	0.727	0.732
Gambusia affinis affinis	Poeciliidae	1	0.909	0.957	0.966	0.924	0.971
Gobio gobio gobio	Cyprinidae	1	0.523	1.000	1.000	0.726	0.927
Lepomis cyanellus	Centrarchidae	1	0.864	0.571	0.857	0.767	0.859
Lepomis gibbosus	Centrarchidae	1	0.477	1.000	0.857	0.767	0.932
Lepomis macrochirus	Centrarchidae	1	0.841	0.846	0.857	0.767	0.928
Lepomis microlophus	Centrarchidae	1	0	0.500	0.857	0.767	0.357
Micropterus salmoides	Centrarchidae	1	0.955	0.829	0.695	0.767	0.936
Mylopharyngodon piceus	Cyprinidae	1	0	0.333	0.333	0.726	0.218
Oncorhynchus mykiss	Salmonidae	1	0.773	0.732	0.510	0.523	0.787
Perca fluviatilis	Percidae	1	0.295	1.000	1.000	0.96	0.956
Rutilus rutilus	Cyprinidae	1	0.455	0.750	0.667	0.726	0.848
Sander lucioperca	Percidae	1	0.091	1.000	1.000	0.960	0.897
Scardinius erythrophthalmus	Cyprinidae	1	0.227	1.000	1.000	0.726	0.894
Tinca tinca	Cyprinidae	1	0.818	0.833	0.833	0.726	0.914
Czechoslovakia							
Clarias gariepinus	Clariidae	0	0.471	0.615	0.714	0.714	0.788
Hucho hucho	Salmonidae	0	0.647	0.000	0.000	0.527	0.269
Ictiobus cyprinella	Catostomidae	0	0.294	0.200	0.357	0.471	0.287
Ictiobus niger	Catostomidae	0	0.294	0.400	0.357	0.471	0.417
Mylopharyngodon piceus	Cyprinidae	0	0	0.417	0.417	0.728	0.265

Introduced species	Family	Outcome[a]	Climate 6	*Prop.species*	*Prop.genus*	*Prop.family*	Probability of establishment
Oreochromis aureus	Cichlidae	0	0	0.880	0.859	0.825	0.667
Oreochromis niloticus	Cichlidae	0	0	0.881	0.859	0.825	0.668
Salvelinus alpinus	Salmonidae	0	0.882	0.781	0.620	0.527	0.836
Salvelinus namaycush	Salmonidae	0	0.706	0.700	0.607	0.527	0.763
Thymallus baicalensis	Salmonidae	0	0	0.000	0.000	0.527	0.050
Ameiurus nebulosus	Ictaluridae	1	0.529	0.946	0.948	0.838	0.940
Aristichthys nobilis	Cyprinidae	1	0.647	0.286	0.286	0.726	0.625
Carassius auratus gibelio	Cyprinidae	1	0.647	0.978	0.962	0.726	0.931
Coregonus lavaretus	Salmonidae	1	0.647	0.714	0.486	0.523	0.754
Coregonus peled	Salmonidae	1	0.647	0.818	0.486	0.523	0.805
Ctenopharyngodon idella	Cyprinidae	1	0.647	0.537	0.537	0.726	0.781
Cyprinus carpio carpio	Cyprinidae	1	0.647	0.907	0.907	0.726	0.916
Gasterosteus aculeatus	Gasterosteidae	1	0.941	1.000	1.000	1.000	0.981
Oncorhynchus mykiss	Salmonidae	1	0.824	0.732	0.510	0.523	0.797
Pseudorasbora parva	Cyprinidae	1	0.647	1.000	1.000	0.726	0.936
Salvelinus fontinalis	Salmonidae	1	0.882	0.639	0.589	0.523	0.769
Sander lucioperca	Percidae	1	1	1.000	1.000	0.960	0.981
Thailand							
Anguilla japonica	Anguillidae	0	0.137	0.200	0.333	0.333	0.121
Mylopharyngodon piceus	Cyprinidae	0	0	0.417	0.417	0.728	0.265
Oncorhynchus mykiss	Salmonidae	0	0.06	0.750	0.520	0.527	0.437
Oncorhynchus rhodurus	Salmonidae	0	0	0.000	0.520	0.527	0.054
Hypophthalmichthys molitrix	Cyprinidae	0	0.009	0.698	0.698	0.728	0.474
Carassius auratus	Cyprinidae	1	0.205	0.978	0.962	0.726	0.878
Cichlasoma octofasciatum	Cichlidae	1	0.094	1.000	0.632	0.822	0.843
Cyprinus carpio	Cyprinidae	1	0.564	0.907	0.907	0.726	0.909
Gambusia affinis affinis	Poeciliidae	1	0.761	0.957	0.966	0.924	0.965
Gymnocorymbus ternetzi	Characidae	1	0.641	1.000	1.000	0.182	0.712

Introduced species	Family	Outcome[a]	Climate 6	*Prop.species*	*Prop.genus*	*Prop.family*	Probability of establishment
Oreochromis aureus	Cichlidae	1	0.53	0.840	0.853	0.822	0.915
Oreochromis mossambicus	Cichlidae	1	0.752	0.940	0.853	0.822	0.948
Oreochromis niloticus	Cichlidae	1	0.838	0.857	0.853	0.822	0.941
Britain							
Coregonus clupeaformis	Salmonidae	0	0	0.000	0.514	0.527	0.054
Hucho hucho	Salmonidae	0	0.237	0.000	0.000	0.527	0.181
Micropterus dolomieu	Centrarchidae	0	0.706	0.308	0.712	0.775	0.703
Oncorhynchus tshawytscha	Salmonidae	0	1	0.267	0.520	0.527	0.583
Oreochromis aureus	Cichlidae	0	0	0.880	0.859	0.825	0.667
Salvelinus namaycush	Salmonidae	0	0	0.700	0.607	0.527	0.304
Ambloplites rupestris	Centrarchidae	1	0.139	1.000	1.000	0.767	0.863
Ameiurus melas	Ictaluridae	1	0.809	1.000	0.948	0.838	0.961
Ameiurus nebulosus	Ictaluridae	1	0.124	0.946	0.948	0.838	0.860
Barbus barbus	Cyprinidae	1	0.892	1.000	0.600	0.726	0.948
Carassius auratus auratus	Cyprinidae	1	0.727	0.978	0.962	0.726	0.936
Cyprinus carpio carpio	Cyprinidae	1	0.902	0.907	0.907	0.726	0.938
Ictalurus punctatus	Ictaluridae	1	0	0.357	0.357	0.838	0.300
Leucaspius delineatus	Cyprinidae	1	0.691	1.000	1.000	0.726	0.938
Leuciscus idus	Cyprinidae	1	0.964	0.500	0.500	0.726	0.830
Oncorhynchus mykiss	Salmonidae	1	1	0.732	0.510	0.523	0.840
Rhodeus sericeus	Cyprinidae	1	0.428	1.000	1.000	0.726	0.922
Salvelinus fontinalis	Salmonidae	1	0.923	0.639	0.589	0.523	0.780
Sander lucioperca	Percidae	1	0.876	1.000	1.000	0.960	0.976
Silurus glanis	Siluridae	1	0.866	1.000	1.000	1.000	0.979
Germany							
Anguilla japonica	Anguillidae	0	0	0.200	0.333	0.333	0.050
Aristichthys nobilis	Cyprinidae	0	0.876	0.310	0.310	0.729	0.704
Coregonus clupeaformis	Salmonidae	0	0.371	0.000	0.514	0.554	0.254

Introduced species	Family	Outcome[a]	Climate 6	*Prop.species*	*Prop.genus*	*Prop.family*	Probability of establishment
Oncorhynchus kisutch	Salmonidae	0	0.343	0.333	0.520	0.554	0.462
Oncorhynchus rhodurus	Salmonidae	0	0	0.000	0.520	0.554	0.059
Oncorhynchus tshawytscha	Salmonidae	0	0.914	0.267	0.520	0.554	0.568
Ameiurus nebulosus	Ictaluridae	1	0.562	0.946	0.948	0.838	0.942
Carassius auratus	Cyprinidae	1	0.914	0.978	0.962	0.727	0.950
Coregonus lavaretus	Salmonidae	1	0.943	0.714	0.486	0.550	0.831
Coregonus peled	Salmonidae	1	0.905	0.818	0.486	0.550	0.861
Cyprinus carpio	Cyprinidae	1	0.943	0.907	0.907	0.727	0.942
Lepomis auritus	Centrarchidae	1	0.543	1.000	0.857	0.767	0.936
Lepomis gibbosus	Centrarchidae	1	0.943	1.000	0.857	0.767	0.960
Oncorhynchus mykiss	Salmonidae	1	0.971	0.732	0.510	0.550	0.845
Pimephales promelas	Cyprinidae	1	0.514	0.667	0.667	0.727	0.822
Pseudorasbora parva	Cyprinidae	1	0.952	1.000	1.000	0.727	0.956
Salvelinus fontinalis	Salmonidae	1	0.962	0.639	0.589	0.550	0.806
Salvelinus namaycush	Salmonidae	1	0.486	0.600	0.589	0.550	0.675
Sander lucioperca	Percidae	1	0.962	1.000	1.000	0.96	0.979
Umbra pygmaea	Umbridae	1	0	1.000	0.800	0.667	0.626
France							
Acipenser baerii	Acipenseridae	0	0.323	0.167	0.059	0.050	0.086
Acipenser ruthenus	Acipenseridae	0	0.165	0.000	0.059	0.050	0.034
Anguilla japonica	Anguillidae	0	0	0.200	0.333	0.333	0.050
Aristichthys nobilis	Cyprinidae	0	0.425	0.310	0.310	0.729	0.596
Clarias gariepinus	Clariidae	0	0.417	0.615	0.714	0.714	0.782
Coregonus clupeaformis	Salmonidae	0	0.173	0.000	0.514	0.554	0.166
Ictalurus punctatus	Ictaluridae	0	0.236	0.429	0.429	0.851	0.698
Oncorhynchus tshawytscha	Salmonidae	0	0.913	0.267	0.520	0.554	0.568
Ambloplites rupestris	Centrarchidae	1	0.772	1.000	1.000	0.767	0.950
Ameiurus melas	Ictaluridae	1	0.701	1.000	0.948	0.838	0.956

Introduced species	Family	Outcome[a]	Climate 6	*Prop.species*	*Prop.genus*	*Prop.family*	Probability of establishment
Ameiurus nebulosus	Ictaluridae	1	0.457	0.946	0.948	0.838	0.936
Carassius auratus	Cyprinidae	1	0.559	0.978	0.962	0.727	0.925
Carassius carassius	Cyprinidae	1	0.984	0.857	0.962	0.727	0.938
Ctenopharyngodon idella	Cyprinidae	1	0.827	0.537	0.537	0.727	0.814
Cyprinus carpio	Cyprinidae	1	0.961	0.907	0.907	0.727	0.944
Gambusia affinis	Poeciliidae	1	0.811	0.957	0.966	0.930	0.967
Lepomis gibbosus	Centrarchidae	1	0.961	1.000	0.857	0.767	0.961
Micropterus salmoides	Centrarchidae	1	0.984	0.829	0.695	0.767	0.939
Oncorhynchus kisutch	Salmonidae	1	0.307	0.222	0.510	0.550	0.369
Oncorhynchus mykiss	Salmonidae	1	0.992	0.732	0.510	0.550	0.850
Pachychilon pictum	Cyprinidae	1	0.126	1.000	1.000	0.727	0.837
Pimephales promelas	Cyprinidae	1	0.283	0.667	0.667	0.727	0.784
Pseudorasbora parva	Cyprinidae	1	0.961	1.000	1.000	0.727	0.957
Salvelinus alpinus	Salmonidae	1	0.661	0.286	0.589	0.550	0.501
Salvelinus fontinalis	Salmonidae	1	0.858	0.639	0.589	0.550	0.777
Sander lucioperca	Percidae	1	0.906	1.000	1.000	0.960	0.977
Umbra pygmaea	Umbridae	1	0.094	1.000	0.800	0.667	0.769
Mexico							
Arapaima gigas	Osteoglossidae	0	0.087	0.000	0.000	0.857	0.226
Aristichthys nobilis	Cyprinidae	0	0.077	0.310	0.310	0.729	0.316
Lepomis microlophus	Centrarchidae	0	0.115	1.000	0.878	0.775	0.846
Salvelinus fontinalis	Salmonidae	0	0.038	0.667	0.607	0.554	0.366
Ambloplites rupestris	Centrarchidae	1	0.106	1.000	1.000	0.767	0.837
Ameiurus natalis	Ictaluridae	1	0.154	1.000	0.948	0.838	0.895
Carassius auratus	Cyprinidae	1	0.404	0.978	0.962	0.727	0.916
Carpiodes cyprinus	Catostomidae	1	0	1.000	1.000	0.412	0.431
Cyprinus carpio	Cyprinidae	1	0.538	0.907	0.907	0.727	0.908
Gambusia affinis	Poeciliidae	1	0.75	0.957	0.966	0.930	0.965

Introduced species	Family	Outcome[a]	Climate 6	***Prop.species***	***Prop.genus***	***Prop.family***	Probability of establishment
Ictalurus punctatus	Ictaluridae	1	0.26	0.357	0.357	0.838	0.657
Lepomis auritus	Centrarchidae	1	0.279	1.000	0.857	0.767	0.916
Lepomis macrochirus	Centrarchidae	1	0.587	0.846	0.857	0.767	0.907
Micropterus dolomieu	Centrarchidae	1	0.298	0.231	0.695	0.767	0.550
Micropterus salmoides	Centrarchidae	1	0.635	0.829	0.695	0.767	0.905
Morone chrysops	Moronidae	1	0.01	0.000	0.300	0.308	0.029
Morone saxatilis	Moronidae	1	0.173	0.200	0.300	0.308	0.134
Oreochromis aureus	Cichlidae	1	0.625	0.840	0.853	0.820	0.922
Oreochromis mossambicus	Cichlidae	1	0.904	0.940	0.853	0.820	0.957
Oreochromis niloticus	Cichlidae	1	0.827	0.857	0.853	0.820	0.940
Poecilia reticulata	Poeciliidae	1	0.692	0.967	0.902	0.930	0.963
Pomoxis annularis	Centrarchidae	1	0.01	0.500	0.571	0.767	0.364
Pomoxis nigromaculatus	Centrarchidae	1	0.221	0.500	0.571	0.767	0.676
Tilapia zillii	Cichlidae	1	0.817	0.882	0.857	0.820	0.943

Values were calculated according to the instructions presented in Section 4.4.
[a]Outcome: 0 = failed; 1 = successful

Table D3. Scores related to risk of establishment in Australia for exotic freshwater fish previously introduced to nine countries[a] calculated using Exotic Freshwater Fish Model 2

Species	Family	Raw Sum Climate 6 Score for Australia	*Prop. species*	*Prop. family*	*S(Climate 6)* for Australia	*Family Random Effect*	Probability of establishment in Australia[b]	Establishment Risk Rank for Australia
Ambloplites rupestris	Centrarchidae	0.304	1	0.77	0.31	-0.045	0.92	Extreme
Ameiurus melas	Ictaluridae	0.16	1	0.84	-0.2	0.04	0.91	Extreme
Ameiurus natalis	Ictaluridae	0.475	1	0.84	0.49	0.04	0.95	Extreme
Gambusia affinis	Poeciliidae	0.834	0.96	0.93	0.85	0.004	0.97	Extreme
Gasterosteus aculeatus	Gasterosteidae	0.246	1	1	0.14	0.0008	0.96	Extreme
Lepomis auritus	Centrarchidae	0.511	1	0.77	0.5	-0.045	0.93	Extreme
Lepomis gibbosus	Centrarchidae	0.244	1	0.77	0.13	-0.045	0.91	Extreme
Lepomis gulosus	Centrarchidae	0.312	1	0.77	0.33	-0.045	0.92	Extreme
Lepomis macrochirus	Centrarchidae	0.694	0.86	0.77	0.7	-0.045	0.92	Extreme
Micropterus salmoides	Centrarchidae	0.875	0.83	0.77	0.9	-0.045	0.93	Extreme
Oreochromis niloticus	Cichlidae	0.597	0.86	0.84	0.6	-0.04	0.93	Extreme
Pseudorasbora parva	Cyprinidae	0.301	1	0.73	0.3	0.13	0.92	Extreme
Sander lucioperca	Percidae	0.152	1	0.96	-0.27	0.013	0.93	Extreme
Scardinius erythrophthalmus	Cyprinidae	0.208	1	0.73	-0.05	0.13	0.9	Extreme
Silurus glanis	Siluridae	0.196	1	1	-0.08	0.002	0.95	Extreme
Tilapia sparrmanii	Cichlidae	0.932	1	0.84	0.92	-0.04	0.96	Extreme
Ameiurus nebulosus	Ictaluridae	0.117	0.95	0.84	-0.5	0.04	0.86	Serious
Barbus barbus	Cyprinidae	0.071	1	0.73	-0.8	0.13	0.8	Serious
Carassius carassius	Cyprinidae	0.217	0.88	0.73	0	0.13	0.86	Serious
Carpiodes cyprinus	Catostomidae	0.017	1	0.44	-1.22	-0.005	0.47	Serious
Catla catla	Cyprinidae	0.161	0.5	0.73	-0.2	0.13	0.63	Serious
Clarias gariepinus	Clariidae	0.931	0.57	0.68	0.92	-0.062	0.82	Serious
Coregonus lavaretus	Salmonidae	0.043	0.75	0.55	-1.03	-0.085	0.41	Serious
Coregonus oxyrhynchus	Salmonidae	0.021	1	0.55	-1.19	-0.085	0.55	Serious

Species	Family	Raw Sum Climate 6 Score for Australia	*Prop. species*	*Prop. family*	*S(Climate 6)* for Australia	*Family Random Effect*	Probability of establishment in Australia[b]	Establishment Risk Rank for Australia
Ctenopharyngodon idella	Cyprinidae	0.222	0.55	0.73	0.07	0.13	0.72	Serious
Esox lucius	Esocidae	0.076	0.89	0.75	-0.78	-0.019	0.73	Serious
Esox niger	Esocidae	0.048	0.5	0.75	-0.95	-0.019	0.42	Serious
Gobio gobio	Cyprinidae	0.197	1	0.73	-0.08	0.13	0.89	Serious
Gymnocorymbus ternetzi	Characidae	0.123	1	0.25	-0.42	0.012	0.52	Serious
Hucho hucho	Salmonidae	0.014	1	0.55	-1.24	-0.085	0.54	Serious
Hypophthalmichthys molitrix	Cyprinidae	0.078	0.68	0.73	-0.77	0.13	0.62	Serious
Ictalurus punctatus	Ictaluridae	0.485	0.4	0.84	0.49	0.04	0.77	Serious
Labeo rohita	Cyprinidae	0.19	0.22	0.73	-0.09	0.13	0.45	Serious
Lates niloticus	Centropomidae	0.144	0.6	0.5	-0.32	-0.016	0.45	Serious
Lepomis cyanellus	Centrarchidae	0.566	0.63	0.77	0.54	-0.045	0.83	Serious
Lepomis microlophus	Centrarchidae	0.285	0.67	0.77	0.27	-0.045	0.81	Serious
Leucaspius delineatus	Cyprinidae	0.148	1	0.73	-0.29	0.13	0.87	Serious
Leuciscus idus	Cyprinidae	0.047	0.67	0.73	-0.97	0.13	0.56	Serious
Macropodus chinensis	Belontiidae	0.093	1	0.81	-0.67	0.007	0.84	Serious
Micropterus dolomieu	Centrarchidae	0.284	0.29	0.77	0.27	-0.045	0.58	Serious
Odontesthes bonariensis	Atherinidae	0.038	1	1	-1.08	0.008	0.87	Serious
Oreochromis macrochir	Cichlidae	0.452	0.33	0.84	0.45	-0.04	0.7	Serious
Pachychilon pictum	Cyprinidae	0.014	1	0.73	-1.24	0.13	0.72	Serious
Pimephales promelas	Cyprinidae	0.296	0.75	0.73	0.29	0.13	0.85	Serious
Pomoxis annularis	Centrarchidae	0.064	0.67	0.77	-0.83	-0.045	0.59	Serious
Pomoxis nigromaculatus	Centrarchidae	0.162	0.6	0.77	-0.19	-0.045	0.69	Serious
Rhodeus ocellatus	Cyprinidae	0.043	1	0.73	-1.03	0.13	0.76	Serious
Rhodeus sericeus	Cyprinidae	0.108	1	0.73	-0.6	0.13	0.83	Serious
Sarotherodon galilaeus	Cichlidae	0.223	0.57	0.84	0.07	-0.04	0.77	Serious
Sarotherodon melanotheron	Cichlidae	0.061	0.67	0.84	-0.84	-0.04	0.64	Serious
Thorichtys meeki	Cichlidae	0.01	1	0.84	-1.27	-0.04	0.75	Serious
Umbra pygmaea	Umbridae	0.031	1	0.71	-1.11	0.024	0.71	Serious

Species	Family	Raw Sum Climate 6 Score for Australia	*Prop. species*	*Prop. family*	*S(Climate 6)* for Australia	*Family Random Effect*	Probability of establishment in Australia[b]	Establishment Risk Rank for Australia
Aristichthys nobilis	Cyprinidae	0.017	0.3	0.73	-1.22	0.13	0.25	Moderate
Barbodes schwanenfeldii	Cyprinidae	0.059	0	0.73	-0.86	0.13	0.16	Moderate
Coregonus peled	Salmonidae	0.004	0.83	0.55	-1.32	-0.085	0.39	Moderate
Gila orcuttii	Cyprinidae	0.062	0	0.73	-0.84	0.13	0.17	Moderate
Lepomis punctatus	Centrarchidae	0.294	0	0.77	0.28	-0.045	0.37	Moderate
Megalobrama amblycephala	Cyprinidae	0.01	0.5	0.73	-1.27	0.13	0.37	Moderate
Morone chrysops	Moronidae	0.099	0.25	0.36	-0.63	0.057	0.13	Moderate
Morone saxatilis	Moronidae	0.39	0.33	0.36	0.39	0.057	0.33	Moderate
Mylopharyngodon piceus	Cyprinidae	0.01	0.38	0.73	-1.27	0.13	0.29	Moderate
Oncorhynchus kisutch	Salmonidae	0.033	0.3	0.55	-1.1	-0.085	0.14	Moderate
Oncorhynchus nerka	Salmonidae	0.063	0.17	0.55	-0.83	-0.085	0.13	Moderate
Salvelinus alpinus	Salmonidae	0.012	0.38	0.55	-1.26	-0.085	0.15	Moderate
Salvelinus namaycush	Salmonidae	0.003	0.64	0.55	-1.33	-0.085	0.27	Moderate
Acipenser baerii	Acipenseridae	0.003	0.14	0.05	-1.33	-0.008	0.02	Low
Acipenser ruthenus	Acipenseridae	0.004	0	0.05	-1.32	-0.008	0.01	Low
Acipenser transmontanus	Acipenseridae	0.065	0	0.05	-0.83	-0.008	0.02	Low
Anguilla japonica	Anguillidae	0.084	0.17	0.31	-0.73	-0.012	0.07	Low
Arapaima gigas	Osteoglossidae	0.006	0	0.75	-1.3	-0.009	0.11	Low
Coregonus autumnalis	Salmonidae	0.013	0	0.4	-1.25	-0.085	0.04	Low
Coregonus clupeaformis	Salmonidae	0.006	0	0.55	-1.3	-0.085	0.05	Low
Esox masquinongy	Esocidae	0.003	0	0.75	-1.33	-0.019	0.1	Low
Ictiobus cyprinella	Catostomidae	0.007	0.17	0.44	-1.3	-0.005	0.07	Low
Ictiobus niger	Catostomidae	0.009	0.33	0.44	-1.28	-0.005	0.1	Low
Oncorhynchus clarki	Salmonidae	0.025	0	0.55	-1.15	-0.085	0.06	Low
Oncorhynchus rhodurus	Salmonidae	0	0	0.55	-1.35	-0.085	0.05	Low
Osphronemus goramy	Osphronemidae	0.02	0.33	0.33	-1.2	-0.007	0.08	Low
Thymallus baicalensis	Salmonidae	0	0	0.55	-1.35	-0.085	0.05	Low

Species	Family	Raw Sum Climate 6 Score for Australia	*Prop. species*	*Prop. family*	*S(Climate 6)* for Australia	*Family Random Effect*	Probability of establishment in Australia[b]	Establishment Risk Rank for Australia
Thymallus thymallus	Salmonidae	0.111	0	0.55	-0.59	-0.085	0.1	Low
Umbra krameri	Umbridae	0.022	0	0.71	-1.18	0.024	0.11	Low

Values were calculated according to the instructions in Section 4.4.

[a]For species not introduced to Australia, but introduced to Italy, Japan, Morocco, Czechoslovakia, Thailand, Britain, France, Germany and/or Mexico.
[b]Probability of establishment = *P(Establishment)* in the instructions in Section 4.4.