Natural Heritage Trust
Helping Communities Helping Australia
A Commonwealth Government Initiative

Australian Government
Bureau of Rural Sciences

Risk assessment model for the import and keeping of exotic reptiles and amphibians

Mary Bomford, Fred Kraus, Mike Braysher,
Liz Walter and Leanne Brown

August 2005

A report produced by the Bureau of Rural Sciences for The Department of Environment and Heritage

---

The Australian Government acting through the Bureau of Rural Sciences has exercised due care and skill in the preparation and compilation of the information and data set out in this publication. Notwithstanding, the Bureau of Rural Sciences, its employees and advisers disclaim all liability, including liability for negligence, for any loss, damage, injury, expense or cost incurred by any person as a result of accessing, using or relying upon any of the information or data set out in this publication to the maximum extent permitted by law.

Postal address:
Bureau of Rural Sciences
GPO Box 858
Canberra, ACT 2601

Internet: http://www.brs.gov.au

Author affiliations:
Mary Bomford: Bureau of Rural Sciences
Mike Braysher: Institute of Applied Ecology,
University of Canberra
Fred Kraus: Bishop Museum, Hawaii
Liz Walter: Braysher Consulting
Leanne Brown: Bureau of Rural Sciences

Foreword

Exotic reptiles and amphibians can establish wild pest populations that cause environmental and economic harm. These introduced species have the potential to cause extinctions of native species or reduce their range and abundance. Their negative impacts on native species can include predation, competition for food, basking sites and other resources, hybridisation and other genetic effects, spread of diseases and parasites, and poisoning through toxic skin glands or venomous bites. Exotic reptiles and amphibians may also alter the habitat of native species and disrupt ecosystem dynamics.

There is a risk that new exotic reptile and amphibian species could establish as wild pests in Australia. If such species escaped or were illegally released into a favourable environment, they could start to breed in the wild and spread to new locations. Once they are widespread, eradication becomes virtually impossible.

Not all exotic reptiles and amphibians pose the same level of threat for establishing a wild pest population. This report addresses the question of whether it is possible to distinguish between species that pose a high risk and those that pose a lower risk. Based on a review of world scientific literature and an analysis of past introductions of exotic reptiles and amphibians to the United States and Britain, it concludes that there is a suite of factors that separates high and low-risk species. This information is used to construct a scientifically based risk assessment model to evaluate the risk that an exotic reptile or amphibian species released into the wild could establish a wild population.

The Bureau of Rural Sciences produced this report for The Department of Environment and Heritage with funding from the Natural Heritage Trust. The report provides information to assist the Australian and State and Territory Governments assess the risks posed by the import and keeping of exotic reptiles and amphibians.

C. Samson

Dr Cliff Samson
Executive Director
Bureau of Rural Sciences

Summary

Over 30% of reported exotic reptile and amphibian introductions have resulted in new populations establishing.

Over 30% of reported exotic reptile and amphibian introductions around the world have resulted in new populations establishing although this figure varies between families and in different locations. This figure may overestimate the true success rate because successful introductions are more likely to be reported.

Assessing invasion risk relies on identifying factors that are linked to the probability of successful establishment if a new exotic reptile or amphibian species was introduced and released in Australia. There is a large scientific literature on the theory of vertebrate invasions, proposing a suite of factors that may influence whether species will establish in new environments.

Establishment risk

There are four key factors for which there is strong evidence of a correlation with establishment success for exotic reptiles and amphibians. These should be considered key factors when the risk that exotic reptile and amphibian species could establish in Australia is assessed:

1. *Number of release events:*

The release of large numbers of reptiles or amphibians at different times and places enhances the chance of successful establishment.

The release of large numbers of animals at different times and places enhances the chance of successful establishment. Small populations are more susceptible to extinction from predation, reduced breeding success, poorer hunting success or increased competition. Chance events such as droughts and floods are also likely to drive small populations to extinction. Small populations may also lose genetic variability, reducing the probability of long-term survival. The minimum viable population size for successful invasion is usually unknown.

An analysis of worldwide introductions of exotic reptiles and amphibians indicates a strong correlation between the number of times a species is introduced and the number of exotic populations it establishes. It appears likely that many reptile and amphibian species can establish exotic populations if sufficient releases are made into suitable environments. This risk can be reduced by restricting which species are kept in Australia, the number of collections holding a species, and the number of individuals held in each collection, and by increasing the security conditions for keeping species and educating people about the risks of releasing exotic reptiles and amphibians.

2. *Climate match*:

Exotic reptiles and amphibians have a greater chance of establishing if they are introduced to an area with a climate that closely matches that of their original range.

Exotic reptiles and amphibians have a greater chance of establishing if they are introduced to an area with a climate that closely matches that of their original range. Species that have a large overseas range over several climatic zones are predicted to be strong future invaders. The suitability of a country's climate for the establishment of a species can be quantified on a broad scale by measuring the climate match between that country and the geographic range of a species. Successfully introduced exotic reptile and amphibian species in the United States and Britain have a greater area of climatically matched habitat than species that were released but failed to establish. Climate matching only sets the broad parameters for determining if an area is suitable for an exotic reptile or amphibian to establish. Other factors, such as the presence of competitors, predators or diseases, might prevent an exotic species from establishing in a climatically matched area.

3. *History of establishing exotic populations elsewhere:*

A history of establishing exotic populations elsewhere is a significant predictor of establishment success.

A history of establishing exotic populations may indicate that a species has attributes that increase the risk of it establishing in other areas. A history of establishing exotic populations elsewhere is a significant predictor of establishment success for exotic reptiles and amphibians introduced to the United States and Britain. 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.

4. *Taxonomic group:*

Taxonomic family is significantly correlated with introduction success rate for exotic reptiles and amphibians.

The introduction success rate for exotic reptiles and amphibians introduced around the world is significantly correlated with a species taxonomic family. The most successful families are: Proteidae; Typhlopidae; Ranidae; Leptodactylidae; Chamaeleonidae; Gekkonidae; Rhacophoridae; Agamidae; Teiidae; Trionychidae; Bufonidae. A precautionary approach is advisable for species that have little or no introduction history, and without relatives with an introduction history.

There are many additional factors listed in the literature that are hypothesized to enhance the probability of establishment for exotic reptiles and amphibians but for which scientific supporting evidence is lacking or equivocal. Rigorously

There are many additional factors listed in the literature that are hypothesized to enhance the probability of establishment but for which scientific supporting evidence is lacking or equivocal.

designed experiments are required to confirm or reject the potential role of these factors:

- a wide geographic range, particularly over extensive continental regions
- high abundance in either their native or introduced range
- an ability to live in human-disturbed habitats (human commensalism)
- broad environmental tolerances
- broad and/or flexible diets (dietary generalists)
- high fecundity (average number of females produced by females surviving to reproductive age) and associated attributes (rapid growth rates and early sexual maturity, large clutch size, frequent breeding, short gestation and opportunistic or aseasonal breeding)
- high genotypic and phenotypic variability and flexibility in behaviour, diet and nesting habits in different environments, because high variability increases the potential for rapid adaptive radiation
- good dispersal abilities
- larger body size
- fit, healthy young animals with low parasite or disease loads
- female able to colonise alone.

Further research is also required to ascertain the importance of:

- care following release, such as sheltering or feeding the newly introduced species
- habitat disturbance in the release site
- an absence or low occurrence of enemies such as predators, competitors, parasites or diseases in the release site

and to confirm the high establishment success of:

- wild caught animals as opposed to captive-reared animals
- animals released on an island as opposed to animals released on the mainland.

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 and human commensalism, this does not mean that it does not influence establishment success. Expert opinion, published in the scientific literature, suggests that such factors may well be potentially important and, thus perhaps they should be considered in the qualitative components of risk assessments.

Scientific theory and knowledge are still inadequate for making certain predictions about the invasive capability of individual species.

Scientific theory and knowledge are still inadequate for making certain predictions about the invasive capability of individual species. This uncertainty has led many experts to question whether it is even feasible to try to reliably predict whether exotic animals could establish in a new country. However, predictions of invasion risk by exotic species based on fairly simple risk assessment models, which include 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. Such simple approaches may overestimate the probability of establishment success, but the low cost of using them to assess risk will enable large numbers of potential invaders to be screened, whereas more complicated approaches require intensive, long-term and expensive study, that make the assessments prohibitively expensive.

Impact risk

The potential impacts of exotic reptiles and amphibians can be classified into three main categories:

1. *Environmental impacts* – including reduced biodiversity, reduced or eliminated endangered species and ecosystem destabilisation.
2. *Economic impacts* – including reduced agricultural productivity or increased production costs, decline in property values, trade effects and damage control costs.
3. *Social and political impacts* – including aesthetic damage, reduced quality of life, consumer concerns and political repercussions.

Unfortunately reliable knowledge about impacts is sparse for most exotic reptiles and amphibians.

Introductions of exotic reptiles and amphibians have often coincided with other changes

Many of the impacts attributed to exotic reptiles and amphibians are correlative or anecdotal. A demonstration of environmental impact requires verification of a causal relationship between changes in a native population or community and the introduction of the exotic reptile or amphibian. Unfortunately reliable knowledge about impacts is sparse for most exotic reptiles and amphibians, both in Australia and overseas, for two main reasons. Firstly, there has been limited research and in particular there are usually scarce pre-invasion data sets. 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 such as habitat disturbance and destruction and the impacts of other introduced species. The combined effects of introduced species and human-caused environmental changes may cause rapid and unpredictable changes in the dynamics of ecological communities.

It is not possible to estimate a reliable figure for the percentage of exotic reptiles and amphibians that become pests because few reliable data on their impacts are available. Hence impacts due to exotic reptiles and amphibians are largely under-reported in the scientific literature.

Competition and predation by exotic reptiles and amphibians both have the potential to be highly detrimental to native species.

Competition and predation by exotic reptiles and amphibians both have the potential to be highly detrimental to native species. Unfortunately scientific knowledge is still inadequate to allow reliable predictions to be made about which exotic species will have the worst impacts due to these factors in new environments. Exotic reptiles and amphibians may also have detrimental effects on recipient ecosystems when they alter the ecological dynamics of natural communities, for example by altering the energy flow through food webs. Diseases spread from exotic reptiles and amphibians to native species may have huge ecological consequences. When exotic reptiles and amphibians hybridise with native species, and produce fertile offspring, the gene pool of the native species is corrupted. This may threaten their survival.

Many species have developed new behaviour when introduced to new environments and hence had impacts that could not have been predicted from their history.

Simple predictions can be made by assuming that invaders will cause significant impacts if they establish in Australia if they have already had negative impacts in other regions. 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. Species that have little harmful effect in their native (or previously introduced) range may have devastating effects when introduced to a new country. Some species have not yet been introduced to new areas, so their pest potential is yet to be demonstrated.

Species that spread rapidly from their initial place of establishment are likely to be harder to eradicate, contain or control. These species are more likely to become widespread and to be considered to be pests than species with a slow rate of spread.

Identifying species that can potentially cause ecological harm is inevitably a subjective process.

Defining harmful species and identifying species that cause or can potentially cause ecological harm is inevitably a subjective process. It is not possible to make reliable decisions about which species are safe to import because they pose a low risk of harm. There is insufficient reliable knowledge of the factors correlated with impacts of exotic reptiles and amphibians to make the development of a quantitative model feasible for assessing the risks of impact for new species of exotic reptiles and amphibians in Australia. Nonetheless, the review of factors associated with adverse impacts above indicates that an increased risk is associated with reptiles and amphibians that:

- 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 for a qualitative assessment of the threat of impacts posed by the establishment of new exotic reptile and amphibian species in Australia. However, an absence of these factors does not indicate a low risk of harm.

The risk of new exotic reptiles and amphibians establishing in Australia can be expected to increase as the number of people keeping exotic reptiles and amphibians increases.

The risk of new exotic reptile and amphibian species establishing in Australia can be expected to increase as the number of people keeping exotic reptiles and amphibians, and the numbers of different reptile and amphibian species kept in collections increases. This is because, as more people keep exotic reptiles and amphibians, the number of escapes and releases of new species is also likely to increase, and establishment of exotic reptiles and amphibians is closely correlated with the number of release events. Any changes to policy or management for exotic species that reduce restrictions on where exotic species can be held, or on the numbers of species held, can increase the risk that more exotic reptiles and amphibians will establish wild populations in Australia. This risk can be reduced by restricting the import and keeping of reptile and amphibian species that:

- have a good climate match to Australia
- have a history of establishing exotic populations elsewhere
- are in a family that has a high introduction success overseas.

A new model presented in this report provides a simple quantitative method for ranking exotic reptiles and amphibians.

A new model presented in this report provides a simple quantitative method for ranking exotic reptiles and amphibians against these factors. The data required to assess reptiles and amphibians species using the model are usually readily available in the literature or on the internet. This model was developed from an assessment of the attributes of exotic reptiles and amphibians that have been introduced to Britain or the United States, using three factors that discriminated between species that successfully established and those that were released but failed to establish wild populations. The model ranks the risk of establishment for an exotic reptile or amphibian at five possible levels, ranging from Very low to Extreme. In this model 85% of the exotic reptiles and amphibians that have established in Britain or the United States are ranked in the Moderate–Extreme risk range and 80% of the exotic reptiles and amphibians that failed to establish are ranked in the Very low–Moderate risk range. Exotic reptiles and amphibians that failed to establish but were ranked as a High or higher risk in the model may establish in the future if there are more releases.

Contents

List of Figures

List of Tables

Introduction

The Bureau of Rural Sciences has developed risk assessment models for the import of exotic mammals, birds and freshwater fish to Australia. The Department of Environment and Heritage required a scientific assessment of whether these models, or a modified version, could be used for exotic reptiles and amphibians. Too few exotic reptile and amphibian species have been released in Australia to enable the necessary scientific analyses to be conducted to assess the existing models. Therefore it was decided to test the components of the risk assessment models on introductions of exotic reptile and amphibian species to overseas countries.

Investigations found by far the best source of up-to-date data on exotic reptile and amphibian introductions around the world is a database developed by Dr. Fred Kraus, Department of Natural Sciences, Bishop Museum, Honolulu, Hawaii. Although this database is not yet complete, its advantages are that it is close to completion, it is restricted to published records of introductions (which makes it more reliable), and it includes records of failed introductions, which are often hard to obtain. Additional species data were added from published literature sources where these were found and judged to be reliable.

Three regions were selected as case studies because they represented both tropical and temperate habitats as well as island and mainland habitats, and they had adequate sample sizes of both successfully established and failed exotic species from a good range of taxonomic families (Table 1).

Table 1. Case study sites and number of exotic reptile and amphibian species that successfully established or failed to establish following introduction.

Case study site	Number of species that established[1]	Number of species that failed to established[1]
Great Britain	8	32
California[2]	11	44
Florida	41	27

[1] Does not include species for which the introduction outcome (successful or failed) is still uncertain.

[2] Includes species translocated from elsewhere in the United States that had no native populations in California.

The model and discussion in this report relates to assessing the potential risks posed by the import and release of exotic reptiles and amphibians. This report does not consider the risks of parasites and pathogens being imported with exotic reptiles and amphibians – this is subject to a separate risk analysis process.

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’. Jurisdictional boundaries are an issue (Shine et al. 2000). 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. For example, the American bullfrog *Rana catesbeiana* has been translocated from its natural range in eastern and central United States to western United States.

Establishment means that a species is breeding in the wild in the country where it has been introduced. The term invasive has no standard definition but is generally taken to mean more than just establishment, and usually indicates an exotic species which spreads well beyond its place of introduction and is also often taken to indicate a species which poses a threat to ecosystems, habitats or native species (Shine et al. 2000). For example, IUCN (2000) defines: ‘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.’

1. Developing reliable predictive risk assessment approaches

Some ecologists doubt that it is possible to build accurate models to predict the outcomes of exotic vertebrate introductions because there has been so little progress in developing generalisations about the factors affecting the establishment and impacts of introduced species (Williamson 1999; Kolar and Lodge 2001). These ecologists often consider the consequences of an introduction are largely dependent on individual circumstances (such as timing, biotic and abiotic components of the invaded habitat, and numbers and condition of introduced animals) and on stochastic events, all of which can make outcomes highly unpredictable. Hence many ecologists claim that predicting the outcome of introductions of exotic invaders will require focused study on each individual potential invader and the recipient habitat (Lodge 1993a, b). Such approaches are costly because they require long-term research. Daehler and Strong (1993), however, suggest the improved predictive power from such expensive studies is questionable because of the poor predictive ability of community ecology even for well-studied systems.

Other ecologists believe that simple models, if based on reliable data, can provide valuable information on invasion threats in the form of robust generalisations, for example: fish invasions in North America (Kolar and Lodge 2002); terrestrial vertebrate invasions in Australia (Bomford 2003); freshwater fish invasions in Australia (Bomford and Glover 2004); and plant invasions (Daehler and Strong 1993). These authors all describe general factors which do not require expensive long-term research and can be used for predictive risk assessments. Predictions from these approaches may not always give complete accuracy regarding invasion success and consequences. However, the low cost of generating these predictions, compared to the potentially high cost of losing endemic native communities and species to exotics, may make them the best available for making decisions on the import and keeping of exotic species.

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 which 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.

The accuracy and consistency of risk assessments, no matter how objective the selection criteria, are dependent on the skill and rigor of the assessor. 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 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 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 these 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 the most qualified and conscientious persons available conduct 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 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 for which a risk cannot be demonstrated due to lack of supporting biological data (Aquatic Nuisance Species Taskforce 1996).

It is important that regulatory agencies take steps to establish and maintain a clear conceptual distinction between assessment of risks and consideration of risk management alternatives. 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 regulatory strategies (Aquatic Nuisance

Species Taskforce 1996). Hence risk managers should not attempt to influence the outcome of an assessment and should ensure that those conducting assessment are free from any pressures or motives that might influence the outcome.

2. 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) followed by 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: human food consumption (9%), intentional introductions for biocontrol (8%), 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. There are four main reasons for exotic species introductions:

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.’ And further 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 (USGS 2003c).

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* 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 (USGS 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 (USGS 2002)

- *Bufo melanostictus* is a large toad widely distributed in Asia but not present in Australia. In the past three years there have been 12 intercepts of *Bufo melanostictus* at the Cairns port (Frank Keenan, pers. comm. 2005). Live individuals have also twice been detected at Darwin dock 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 nearby Java 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. Reasons for release

Releases may be of unwanted pets, or people intentionally trying to establish wild populations, or accidental escapes. The following are a few examples from the literature:

- Common methods of release for exotic reptiles and amphibians in Florida include release of pets, escape from pet dealers, or intentional release for pest control (King and Krakauer 1966, cited in Wilson and Porras 1983).

- King and Krakauer 1966, cited in Wilson and Porras 1983) listed 2361 individuals of 17 species of reptiles and amphibians as being released in 1964 at an address of an animal dealer in Florida.

- Spinks et al. (2003) captured nine non-native turtle species in a waterway at the University of Davis, California. 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 trade. Spinks et al. (2003) suggest 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 *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 juvenile *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 10 cm in length are not legally available within the pet trade in the United States.

- *Rana catesbeiana* has 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).

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

- There have been widespread releases of red-eared sliders *Trachemys scripta* 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 and the species is now widely distributed in France. According to Cadi and Joly (2004), more than 52 million red-eared slider individuals 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.

- *Ambystoma tigrinum* has been deliberately introduced as fish bait in central throughout the American West (Riley et al. 2003).

- Bullfrog *Rana 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).

- Some species, such as the cane toad (*Bufo marinus*) were introduced intentionally as agents of biocontrol around the world (Easteal 1981; USGS 2003d). 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, U.S. Virgin Islands, the Territories of Guam, American Samoa, and Australia (McCoid 1995).

4. Reasons for intentional or assisted spread

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

- Spread of *Eleutherodactylus coqui* and *E. planirostris* 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 *Bufo marinus* toads and release them in their back yards 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 bullfrogs *Rana catesbeiana* may have been accidentally spread in western United States during fish-stocking operations and that people may have also intentionally spread them for hunting.

5. Control and eradication

The release of a few individuals can lead to a rapidly expanding population. 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 for eradication to be attempted 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). *Xenopus 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, *Xenopus laevis* aestivates in underground fissures (Tinsley and McCoid 1996). The species is difficult to control. Poisoning using a range of chemicals including high concentrations of Rotenone failed to eradicate it in California and has even failed to prevent population expansion (Tinsley and McCoid 1996; Lafferty and Page 1997). Chemical controls are likely to have undesirable effects on native species. While trapping is safer it is labour intensive and thus expensive. All adults in a population are unlikely to be trapped and this method cannot be used for tadpoles or eggs.

Section 1: Review of factors affecting the potential of an exotic reptile or amphibian to establish a free-living population in the wild

1.1 Establishment success rates

According to some ecologists, only about 10% of exotic introductions to the wild succeed in establishing (Williamson 1996, 1999; Williamson and Fitter 1996; Holmes 1998; Ricciardi and Rasmussen 1998; Enserink 1999; Smith et al. 1999). Analyses of past introductions of exotic birds, mammals and fish reveal that this generalisation is doubtful for vertebrates with success rates varying between around 30–40% for birds (Bomford 2003), 51% for exotic freshwater fish (Arthington et al. 1999; Bomford and Glover 2004) and around 65–80% for mammals (Bomford 2003). Data collated by F. Kraus on published records of introduction outcomes of exotic and translocated reptiles and amphibians introduced around the world indicate that 31% of recorded introduction events have resulted in successful establishment. These data may be biased in favour of successful introductions. If more unsuccessful introductions were recorded, the reported success rates could be lower (Welcomme 1988; Moyle and Light 1996b; MacIsaac et al. 2001). Unsuccessful introductions may only be reported for easy to detect species or species that are frequently released or that are released in well-surveyed sites.

1.2 Predicting establishment success

Assessing the risk of establishment for an exotic reptile or amphibian species released in Australia relies on identifying the factors that are correlated with establishment success. There is considerable scientific literature on the ecological theory of species' invasions, proposing a suite of factors that may influence whether or not exotic vertebrates establish in new environments. These factors are evaluated below with an assessment of their practical significance for predicting the risk of new reptile or amphibian species establishing in Australia.

1.2.1 Key factors that are predictive for establishment success

There are four key factors for which there is strong evidence of a correlation with establishment success:

(i) Introduction effort – numbers of animals released and number of places and times at which releases occur

The release of large numbers of animals at different times and places may enhance the chance of successful establishment for freshwater fish, mammals and birds (Duncan et al. 2001; Bomford 2003; Bomford and Glover 2004; Forsyth et al. 2004). It is likely the same relationship holds true for reptiles and amphibians. Small populations are more susceptible to extinction from predation, reduced breeding success, poorer hunting success or increased inter-specific competition. Chance events such as droughts and floods are also likely to drive small populations to extinction. Small populations may also lose genetic variability that may reduce the probability of long-term survival. The minimum viable population size for successful invasion is not known for most species.

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.

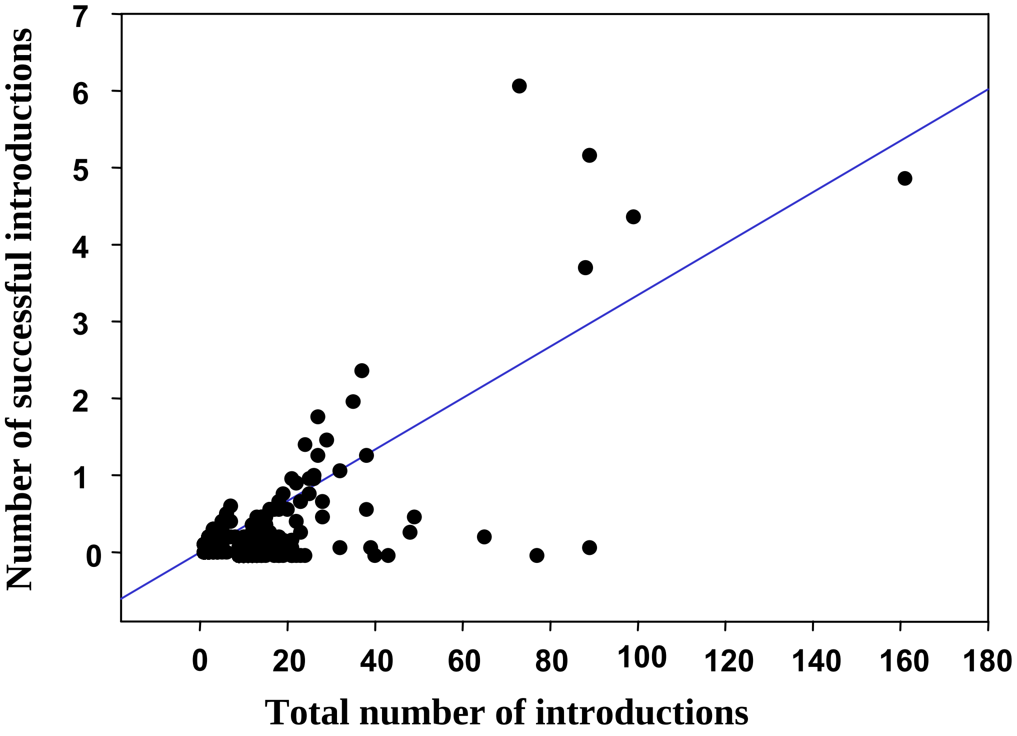

Figure 1. Relationship between the number of introduction events and the number of successful establishment events, for exotic and translocated reptile and amphibian species introduced around the world. Introduction events include multiple introductions to a jurisdiction (countries or individual states in North America). (Data from F. Kraus unpublished database of 1700 records).

Kraus (based on updated database of 1700 records) found a strong relationship between the number of introduction events and the number of successful establishment events, for reptile and amphibian species introduced around the world (Figure 1). These data show clearly that species that have been introduced more frequently have established more exotic populations, indicating that introduction effort has a strong influence on establishment success.

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 *Trachemys 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 (Salzberg1998; Spinks et al. 2003).

For some species, even a small propagule or a single individual may be sufficient to found an exotic population (see Section 1.2.2 vi). There are many examples of exotic populations starting on small islands from small introduction propagules. An introduction of seven individuals of *Lacerta sicula* (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 *Lacerta melisellensis* population (Nevo et al. 1972). Losos et al. (1997) introduced populations of *Anolis sagrei* onto small islands from a nearby source. They introduced propagules of 5 or 10 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 *Rana catesbeiana* in the Netherlands. The population started from the release of five bullfrog tadpoles in 1986 into a newly constructed garden pond.

Invasion via multiple loci is the most effective means of establishing exotic species in new environments. For example, *Eleutherodactylus coqui* and *E. planirostris* 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, 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 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; and by educating people about the risks of releasing exotic reptiles and amphibians. 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 similarity between the sites of origin and release based on rainfall and temperature data. Potential species' ranges are predicted using a 'climate envelope' approach, in which the current distribution of a species is mapped and its climatic attributes measured, and then extralimital locations with matching climate attributes are determined and mapped. The expectation is that a species is likely to be able to establish in locations with a climate closely matched to that in its current range. Climate match is a significant predictor of introduction success for birds, mammals and freshwater fish (Duncan et al. 2001; Forsyth et al. 2004; Bomford and Glover 2004). The same relationship holds true for reptiles and amphibians introduced to Florida, California and Britain (Figure 2).

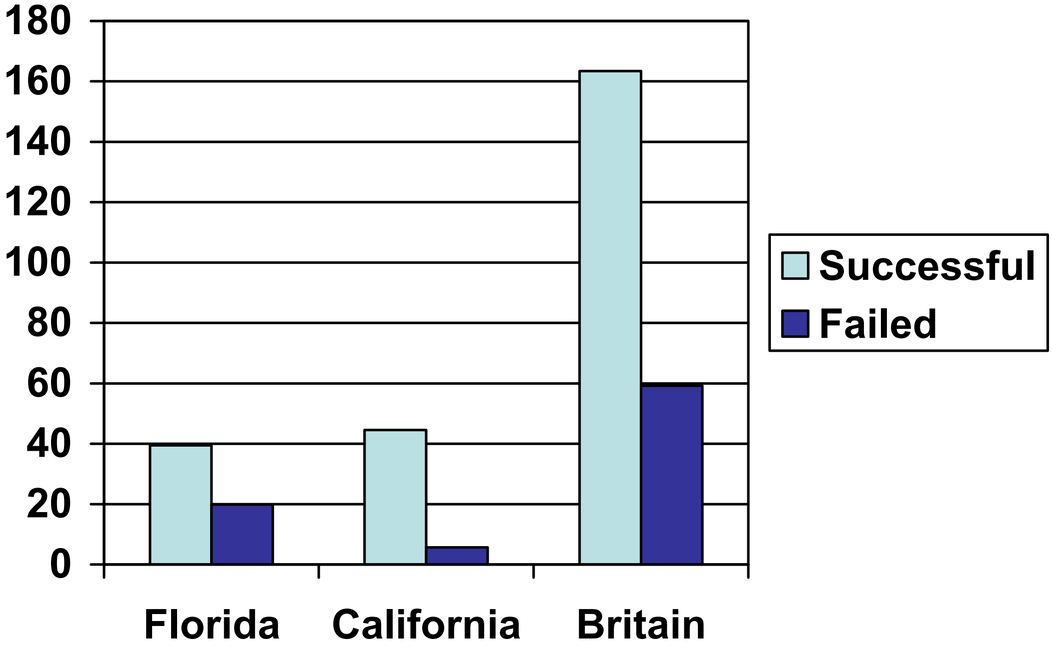

Figure 2. Climate match scores (Euclidian Σ7–10) for exotic reptiles and amphibians introduced to Florida, California and Britain. The differences for successfully established and failed species are statistically highly significant for all three jurisdictions. (Data and analysis presented in Appendix A.)

Freezing weather can eliminate newly released propagules of reptiles and amphibians if 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 *Eleutherodactylus coqui,* a small frog native to Puerto Rico, was eradicated by freezing weather (Wilson and Porras, 1983). Wilson and Porras (1983) also suggest freezing weather can extirpate populations of newly established exotic frogs such as *Eleutherodactylus coqui* 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*, *Coronella austriaca* and *Natrix natrix*.

There has been widespread release of red-eared sliders *Trachemys 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 native *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.

Xenopus laevis is principally confined to aquatic habitats though able to move overland between water bodies (Measey 1998c; 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 *Xenopus 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 and limited to warmer months and occurs at one-third the rate of *X. laevis* in California (Measey 1998c). 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 1998c)

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 Britain 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 in Australia. The climate match between a species' overseas geographic range and mainland Australia can be determined using CLIMATE software. Species with a high climate match to Australia are most likely to establish here. 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, so that 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 in Australia.

(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). Species of exotic freshwater fish, mammals and birds introduced to Australia that have a history of establishing exotic populations elsewhere are more likely to establish exotic populations here (Duncan et al. 2001; Forsyth et al. 2004; Bomford and Glover 2004). The same relationship holds true for reptiles and amphibians introduced to Florida, California and Britain (Figure 3).

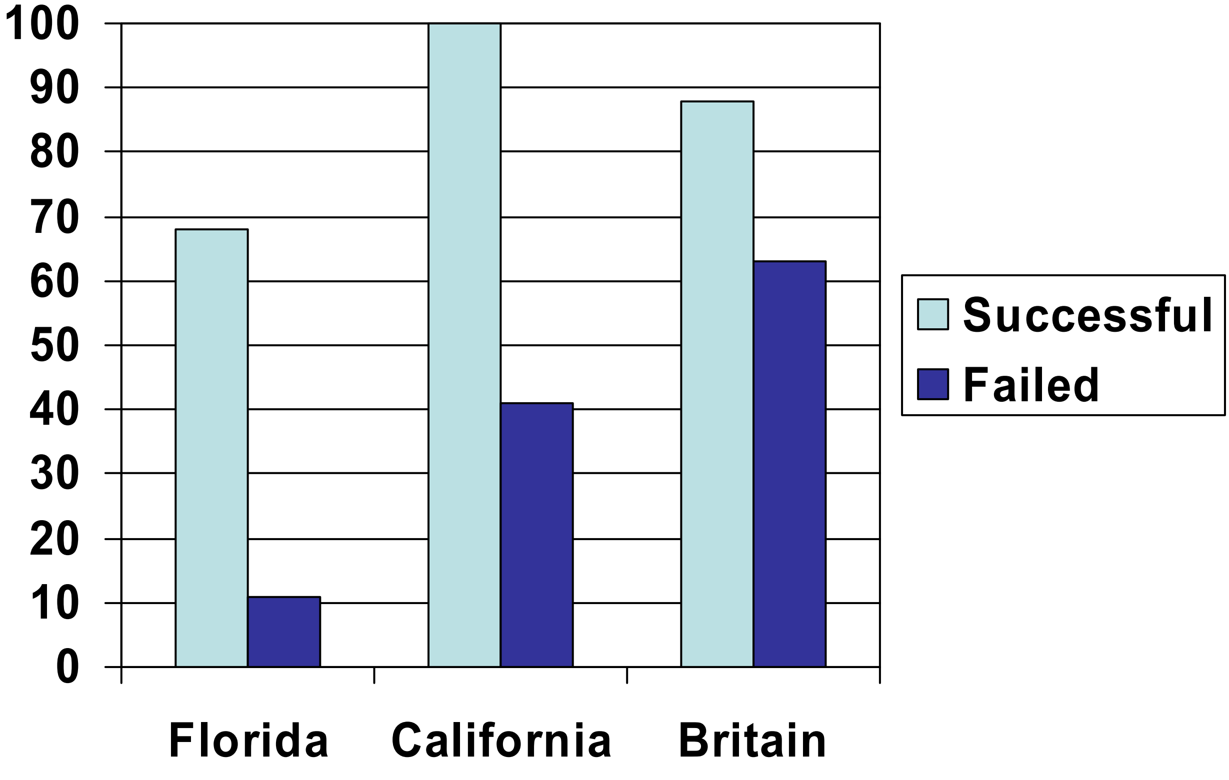

Figure 3. Percentages of exotic reptiles and amphibians introduced to Florida, California and Britain that have established exotic populations elsewhere. Species that successfully established in each jurisdiction (left-hand columns) were more likely to also have successfully established elsewhere than species that failed to establish in that jurisdiction (right-hand columns). Data presented in Appendix A.

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, this variable should be considered as a key factor when assessing the risk that exotic reptiles and amphibians 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, 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 some vertebrate bird, mammal and freshwater fish families have higher establishment success rates than others (Bomford 2003; Bomford and Glover 2004). This relationship is also true for exotic reptiles and amphibians (Table 2).

Table 2. Number of introductions (to countries or to states/provinces for North America) for exotic or translocated reptile and amphibian species in the listed families and the number and percentage of these introduction events that resulted in successful establishment. Success rates vary from 100% to zero%. (Data sourced from F. Kraus, unpublished database).

Family	Number of introduction events	Number of successful establishment events	% Successful
Dendrobatidae	1	1	100
Proteidae	7	7	100
Typhlopidae	65	62	95
Ranidae	143	114	80
Leptodactylidae	53	42	79
Chamaeleonidae	14	11	79
Gekkonidae	256	195	76
Rhacophoridae	4	3	75
Agamidae	20	14	70
Teiidae	9	6	67
Trionychidae	38	25	66
Bufonidae	87	52	60
Microhylidae	10	6	60
Plethodontidae	12	7	58
Lacertidae	51	29	57
Iguanidae	148	83	56
Testudinidae	42	20	48
Scincidae	57	26	46
Pipidae	19	8	42
Hylidae	87	36	41
Myobatrachidae	5	2	40
Emydidae	193	73	39
Discoglossidae	13	5	38
Ambystomatidae	13	5	38
Varanidae	13	5	38
Salamandridae	25	9	36
Anguidae	7	2	29
Chelydridae	14	4	29
Pelomedusidae	12	3	25
Chelidae	9	2	22
Viperidae	24	5	21

Colubridae	133	26	20
Cordylidae	6	1	17
Alligatoridae	20	3	15
Elapidae	18	2	11
Boidae	49	3	6
Pelobatidae	2	0	0
Cryptobranchidae	2	0	0
Amphisbaenidae	1	0	0
Gymnophthalmidae	1	0	0
Helodermatidae	3	0	0
Pygopodidae	1	0	0
Kinosternidae	5	0	0
Crocodylidae	2	0	0
Geomydidae	1	0	0

Risk assessment significance: For reptile and amphibian species with a history of introductions to new areas, or with con-familial relatives having such a history (listed in Table 2), 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

1.2.2 Other factors that may influence establishment success

There are ten additional factors which may influence establishment success but for which supporting data are lacking:

(i) Overseas geographic range size

Species that are widespread and abundant in their original range, particularly over extensive continental regions, may be more likely to establish exotic populations. This relationship holds for exotic freshwater fish, mammals and birds introduced to Australia (Bomford 2003; Duncan et al. 2001; Bomford and Glover 2004; Forsyth et al. 2004). However, it appears to be of no or only minor significance for exotic reptiles and amphibians introduced to Florida, California and Britain (Figure 4).

Campbell and Echternacht (2003) suggested the extensive native range of the brown anole *Anolis sagrei* is one of the characteristics that contributes to its successful invasion history

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.

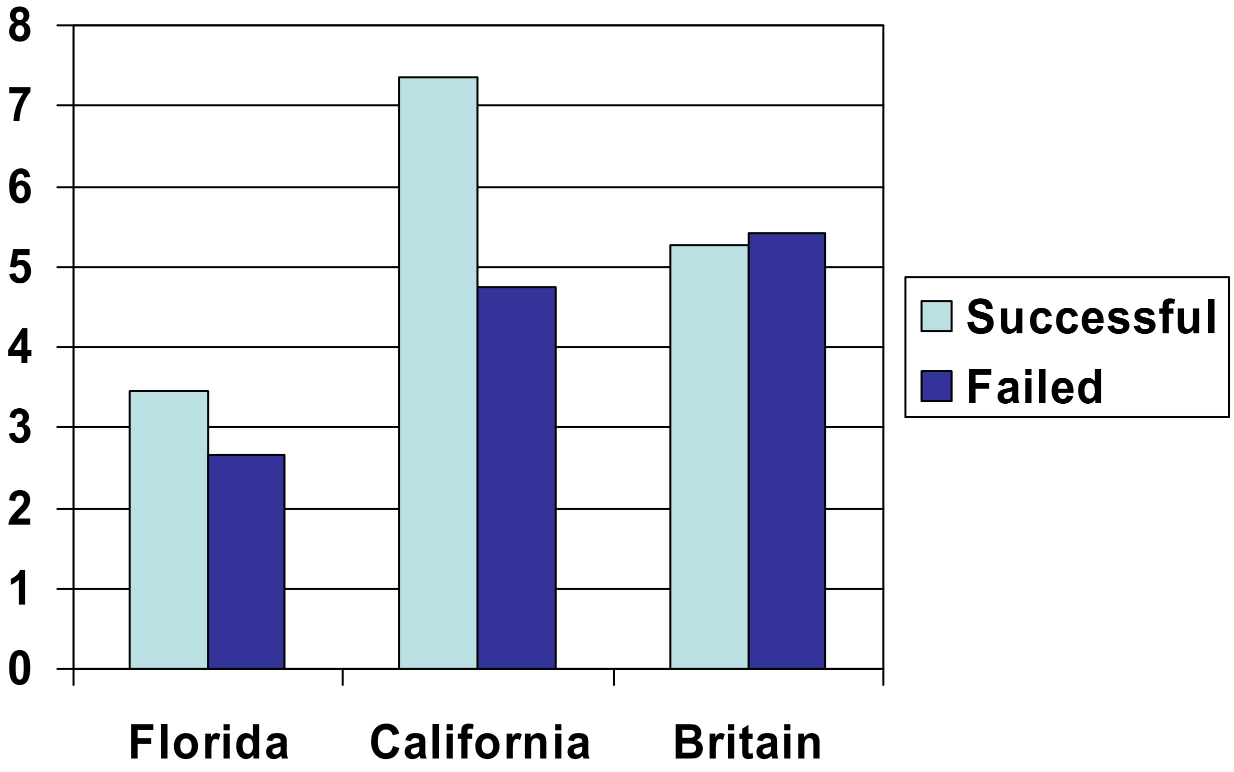

Figure 4. Average geographic range sizes (millions of square kilometres) for exotic reptiles and amphibians introduced to Florida, California and Britain. The differences for successfully established and failed species are not statistically significant. Data presented in Appendix A.

(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 Britain 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 have since established in natural areas – these include *Eleutherodactylus planirostris*, *Osteopilus septentrionalis* and *Anolis sagrei* (Butterfield et al. 1997) and *Python molurus* (Fred Kraus pers. obs.)

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 *Rana 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 *Xenopus 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 *Bufo marinus*. Wilson and Porras (1983) found that the Cuban treefrog *Osteopilus 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 *Anolis 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 *Rana catesbeiana* is facilitated by the presence of co-evolved non-native fish, which increase tadpole survival by reducing predatory macroinvertebrate densities. Native dragonfly nymphs in Oregon, 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 then 'set the stage' for invasion by coqui and other exotic frog species.

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 in Australia. However, it is necessary to recognise that while environmental disturbance may enhance probability of success, it is also possible for exotic reptiles and amphibians that can live in disturbed environments to 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. Case and Bolger (1991a, b) 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* species are present on both Guana Island and on Guam in the Pacific. They also found that the failure of introduced *Hemidactylus mabouia* to proliferate away from human habitation on Guana Island was unrelated to the presence of native lizard competitors (nocturnal predators) since none is known from the island. This example suggests caution in invoking competition to explain the abundance or distribution of *Hemidactylus 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 *Anolis porcatus* occurred, its ecological analogue, the native *Anolis chlorocyanus* was uncommon or absent and conversely, where *Anolis chlorocyanus* was common, *Anolis porcatus* was apparently absent. Introduction in a locality of *Anolis porcatus* led to a decline in *Anolis 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 *Anolis equestris* could hinder the establishment of exotic *Anolis 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 Guam. Their modelling suggests that prey abundance both on Guam and in native range of *Boiga 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 *Boiga irregularis*. Based on this estimation of environmental suitability, the authors predict that *Boiga 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 *Boiga irregularis* is attributable to the success on Guam of introduced prey species, especially the house gecko *Hemidactylus frenatus* and the terrestrial skink *Carlia ailanpilai*. Other important prey items are introduced birds, especially chickens *Gallus gallus*, francolins *Francolinus francolinus*, drongos *Dicrurus macrocercus*, tree sparrows *Passer montanus*, rock doves *Columba livia*, turtle doves *Streptopelia bitorquata*, 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 *Boiga irregularis* and the irruptions of shrews *Suncus murinus* and skinks *Carlia ailanpilai* accelerated the population expansion of *Boiga irregularis*. Were it not for the highly successful introduced prey species Guam would probably not now have a dense population of brown treesnakes.

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 receptor 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, 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 these biotic components of habitat suitability objectively. 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) suggests that being a generalist contributes to the ability of *Hemidactylus frenatus* to colonize locations outside its natural range. Da Silva and Blasco (1995) consider it likely that the broad ecological tolerances and omnivorous diet of *Trachemys 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* 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, treefrogs, smaller anoles, young birds, young rodents. Campbell and Echternacht (2003) suggest that the generalised diet of the brown anole *Anolis 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 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 *Anolis sagrei* in southeast Florida is its broad adaptability in edificarian areas – that is, habitats dominated by buildings, with little vegetation. Wilson and Porras (1983) also suggest that one reason for the success of *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 its native range and its generalized habitat use are two of the characteristics that contribute to the successful invasion history of *Anolis sagrei*. 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 females produced by females surviving 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 *Anolis equestris* in urban areas of south Florida may be its longevity relative to most other anoline species. According to McCoid and Fritts (1993, 1995) *Xenopus laevis* in California has 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 for predicting the probability of establishment success at present.

(vii) Single female able to colonise alone

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

A number of lizard species, like *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 *Anolis sagrei*.

Risk assessment significance: Although there is no evidence to support the theory that species in which a single individual can colonise alone have higher introduction success, it is possible that for such species the minimum viable propagule size is lower (see Section 1.2.1 i), and that therefore there may be a higher risk of establishment for such species.

(viii) Dispersal ability

Da Silva and Blasco (1995) suggest it is likely that the dispersal ability of *Trachemys scripta* 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 *Anolis 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 and 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. This position may change if new evidence becomes available.

(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), but 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, and 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).

Cole et al. (2005) suggests that small body size contributes to the ability of *Hemidactylus frenatus* to colonize 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. This position may change if new evidence becomes available.

1.3 Discussion

There is strong evidence that the four key factors listed in Section 1.2.1 are correlated with establishment success for introduced exotic reptiles and amphibians. A good climate match, and a history of establishing exotic populations elsewhere are statistically significantly associated with establishment success for Britain, Florida and California. There is also a significant correlation between number of release events and number of establishment events for exotic reptiles and amphibians introduced around the world. Taxonomic family is also significantly correlated with establishment success. When conducting these types of statistical analyses it is desirable to control for the degree to which the reptiles and amphibians in the samples are genetically related to one another as this can bias the results. This requires a detailed understanding of the systematics of reptiles and amphibians so that phylogenetically corrected analyses can be conducted. Although this was not feasible within the constraints of the current study, it is certainly desirable as a future project. However, the correlations we demonstrated are considered to form a valid basis for developing a model to predict the risk posed by future introductions of exotic reptiles and amphibians to Australia. A simple quantitative model to predict the risk of new species of exotic reptiles and amphibians establishing in Australia is developed in

Section 3 of this report.

The factors listed in Section 1.2.2 have not been confirmed to be correlated with establishment success. As might be expected from such a taxonomically diverse group as reptiles and amphibians, successfully introduced species differ widely from one another in breeding habits, degree of parental care, adult size, feeding habits, dispersal ability, environmental tolerances and many other factors. Rigorously designed experiments are required to confirm or reject the potential role of these factors. Further assessment of these factors was beyond the scope of this project and more research would be required to establish their significance. In the meantime, these factors should probably be considered in qualitative assessments of risk, in addition to the quantitative risk assessment approach developed in Section 3 of this report. This would be particularly desirable if decisions are being made on whether to import species of exotic reptiles and amphibians that score a moderate or higher risk of establishment in the quantitative risk assessment model.

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 the niche it occupies in its native range.

Moyle and Light (1996b) considered the abiotic conditions of the environment (in this study measured by climate match) in relation to the biological characteristics of the invader are by far the most important factors determining invasion success. They considered biotic resistance (competitors, predators diseases etc) far less significant, although they acknowledged predation may be important in the initial stages of an introduction if the numbers of the invading reptiles and amphibians are low. In contrast, Lodge (1993a) considers that the role of predation, competition, disease and other subtle interactions are perhaps just as important as the abiotic conditions but are underestimated because they are more difficult to measure.

Scientific theory and knowledge are still unable to be used to make certain predictions about the invasive capability of individual species. This uncertainty has led many experts to question whether it is even feasible to try to reliably predict whether exotic animals could establish in a new country (Crawley 1989; Ehrlich 1989; Williamson 1989, 1996; Gilpin 1990; di Castri 1991; Fryer 1991; Lidicker 1991; Norton et al. 1996; Ricciardi and Rasmussen 1998; Arthington et al. 1999; Enserink 1999). Williamson (1996) concludes that an invader can be any sort of species going into any sort of environment. Some experts believe that current ecological theory on animal invasions is inadequate to make quantitative scientific predictions (Crawley 1986, 1989; Brown 1989; Simberloff 1989). Ehrlich (1989) stated that ‘One certainty is that population biologists are still a long way from any comprehensive quantitative theory of what determines the potential for becoming a successful invader’. He suggests that such a theory may not be possible because demographic and environmental stochasticity plays such a large part in any individual introduction that it is not possible to generate mathematical probability distributions of likely success. Nevertheless, he points out that, despite these high levels of uncertainty, what is known is far from trivial, and ecological knowledge can contribute much to assessing the probability of invasion success.

Daehler and Strong (1993) suggest that predictions of invasion risk by exotic species, based on fairly simple risk assessment models will allow predictions to be made at low-cost to guide management policies. Daehler and Strong (1993) acknowledge that simple approaches may overestimate the probability of establishment success, but consider their simplicity and low cost will enable large numbers of potential invaders to be screened. Whereas more complicated approaches that require intensive, long-term and expensive study of the biology and ecology of introduced species and the structure and function of potentially invaded ecosystems, will preclude the assessment of many species. While acknowledging the potential value of such simple approaches, Lodge (1993c) queried their reliability, because he considered that the characteristics of the community being invaded are as critical to establishment success of an introduced exotic species as the characteristics of the introduced species. However, Moyle and Light (1996a, b) contend that all aquatic systems are invasible regardless of the biota already present, if abiotic conditions are appropriate. If this is true, fairly simple modelling involving climate matching and past history of invasion, should go much of the way to determining the risk of establishment posed by introduced species, at least over broad geographic areas.

While Erlich's (1989) contention that there is no comprehensive quantitative theory for determining which species will become successful invaders is still essentially true, in the last decade several papers have demonstrated statistically significant links between many of the factors listed in Section 1.2 and establishment success for exotic vertebrates. This information can be used to give probabilistic estimates of whether an exotic species released in Australia is likely to successfully establish. Unfortunately, these studies have not included reptile and amphibian species. Hence, where there is no evidence presented in this report in support of the role of some factors, such as for diet or human commensalism, this does not mean that these factors do not influence establishment success. Expert opinion, published in the scientific literature, suggests that such factors may be important, and thus perhaps they should be considered in qualitative components of risk assessments.

Factors associated with establishment success are not necessarily the same as the factors associated with spread following establishment (Duncan et al. 2001; Kolar and Lodge 2002; Forsyth et al. 2004). For example, Kolar and Lodge (2002) found relatively fast body growth was positively associated with establishment for fish but was negatively associated with fast spread. Duncan et al. (2001) and Forsyth et al. (2004) found that exotic birds and mammals that have spread widely in Australia have traits associated with faster population growth rate (including small body size, shorter life span, lower weaning age, short incubation periods, more offspring or broods per year) as well as having larger overseas range sizes and a better climate match than species that have not become widespread. It would appear that traits associated with spread may be taxon-specific or location-specific. However, Kolar and Lodge (2002) examined traits associated with rate of spread, whereas Duncan et al. (2001) and Forsyth et al. (2004) examined traits associated with extent of spread, which could account for the major differences found in these studies. No information has been collated or assessed on the factors associated with rate of spread or area of spread for exotic reptiles and amphibians.

Section 2: Review of factors affecting the potential impacts of exotic reptiles and amphibians

2.1 Types of impact

The potential impacts of exotic reptiles and amphibians can be classified into three main categories:

1. *Environmental impacts* including: ecosystem destabilisation, reduced biodiversity, reduced or eliminated keystone species, reduced or eliminated endangered or threatened species; effects of control measures (an indirect effect).
2. *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; injuries to people or domestic animals.
3. *Social and political impacts* including: aesthetic damage; health effects; reduced quality of life; consumer concerns; political repercussions.

The environmental impacts of exotic reptile and amphibian species 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 assessment 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 endangered species and biodiversity
- Ecosystem disturbance 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). Geographically or evolutionary isolated ecosystems, 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 every exotic species needs to be treated for management purposes as potentially invasive, unless or until there is reasonable indication that this is not so. They assert that this is why the precautionary principle, based on scientific evidence, should underpin all preventative legal frameworks.

Exotic reptiles and amphibians may also have positive impacts, for example, as a biocontrol agent, pet or display specimen, or use for food production (for example edible frogs).

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 and interest groups 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. 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 reptile and amphibian species 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 changes in a native species' population or a natural community and the presence of an exotic reptile and amphibian. 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 reptiles and amphibians. 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 reptiles and amphibians often occurs concurrently with other changes which 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. It may also be possible to demonstrate impact following an introduction, by experimentally manipulating densities of the exotic species and monitoring community responses.

The best method for developing a predictive model for the impact of reptile and amphibian 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. It would then be possible to look at the attributes of reptiles and amphibians with known impacts, to determine any attributes associated with harmful impacts. Unfortunately, for most known reptile and amphibian invaders, 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 (Section 2.3) 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 reptiles and amphibians (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).

2.3 Reliability of evidence and state of knowledge on impacts

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; Smith2005, 2005 in press). 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 their introduction 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 in particular there are usually scarce preinvasion data sets. Secondly, introductions of exotic reptiles and amphibians have often coincided with other changes which means impacts due to exotic reptiles and amphibians are confounded with impacts due to other factors, making it difficult to determine the impacts of exotic species introductions. Factors that may be confounded with the impacts of exotic reptiles and amphibians include:

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

- The introduced red-eared slider *Trachemys 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 *Rana 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), introduced *Bufo marinus* is replacing the native southern toad *Bufo terrestris* in Florida and this has sometimes been attributed to competition between the two species. But Wilson and Porras (1983) suggest that *Bufo 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 *Bufo marinus*.

- In Papua New Guinea, the Papuan black snake *Pseudechis papuanus* apparently declined around Port Moresby following the introduction of the cane toad *Bufo 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 *Xenopus 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 including: converting coastal wetlands to marinas; highway and roadway construction; freshwater diversions; grazing, breaching of coastal lagoons; and flood control practices.
- According to Spinks et al. (2003), habitat destruction, human disturbance, irrigation and exotic predators are all responsible for increasing mortality of native *Actinemys marmorata* populations in California. Hence it is difficult to separate the effects of these impacts from the effects of competition and predation by introduced *Trachemys scripta*.

(ii) Impacts of other introduced animals

- Although introduced bullfrogs *Rana catesbeiana* have 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 *Rana catesbeiana* is absent.
- Adams et al. (2003) found that invasion by introduced bullfrogs *Rana catesbeiana* in western North America is facilitated by the presence of a co-evolved non-native fish, which increase bullfrog tadpole survival by reducing predatory macroinvertebrate densities.
- Luiselli et al. (1997) suggest the impacts of the introduced red-eared slider *Trachemys scripta leprosa* on the native pond turtle *Emys orbicularis* in Europe, may be confounded by the presence of other introduced pond turtles including *Mauremys cascipa* and *Mauremys 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.
- According to Lever (2003), 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 *Emoia atrocostata* and the mottled snake-eyed skink *Cryptoblepharus poecilopleurus* in the following decades. However, the Asian musk shrew *Suncus murinus* was introduced at the same time as *Carlia 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 *Xenopus 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 introduced curious skink *Carlia ailanpalai* and introduced brown tree snake *Boiga 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 introduced bullfrogs *Rana catesbeiana* have been blamed for the decline of the California red-legged frog *Rana 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.

According to some ecologists, only about 10% 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 around 50% become pests (Bomford 2003; Bomford and Glover 2004). It is not possible to estimate a reliable figure for the percentage of exotic reptiles and amphibians that become pests because few reliable data on the impacts of exotic reptiles and amphibians 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).

2.4 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 islands in central Florida (forested and non-forested). 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, although an alternative explanation is that food resources may be more abundant. 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, such as 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 and 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 *Anolis sagrei* 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 different the recipient island's vegetation was from the vegetation where the lizards were sourced, the greater the magnitude of the differentiation.

Some exotic species spread quickly whereas others 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 and interactions with it (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 *Hemidactylus frenatus* spent over a century in Australia confined to a few local footholds largely commensal with human settlements. It has since spread widely and in a few decades expanded its range 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 and in others geographical barriers (such as being on an island) has 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, *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.

2.5 Types of impacts and their significance for impact risk assessment

A review of the literature on exotic reptiles and amphibians introductions indicates the following types of impact may occur. These are briefly described, together with examples and their risk assessment significance.

2.5.1 Competition for resources

Interspecific competition can lead to reduced growth rates, survival and recruitment 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, aggression or the release of toxins. In exploitation competition, competitors differ in their ability to exploit resources.

Anurans

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 growth inhibition 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. 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 2 of 13 assays and the magnitude of the inhibition was much less than for laboratory experiments with crowded tadpoles. So Petranka (1989) concluded interference competition involving growth inhibitors can occur but appears to be 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.

(i) American bullfrog *Rana catesbeiana*

There is strong field observational and experimental evidence that bullfrogs *Rana catesbeiana,* introduced to western United States from the eastern states, compete with native ranid frogs such as *Rana pretiosa, R. pipiens, R. draytonii, R. aurora* and *R. boylii* for resources (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. 2001a). 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 *Rana 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 no significant effect on survival. Responses to bullfrogs in field settings were similar qualitatively 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 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 and 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 *Rana 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 33%. 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, although predation possibly contributed, as bullfrog tadpoles eat tadpoles of other species, including *R. aurora* in the laboratory.

Behavioural observations by Kiesecker et al. (2001a) 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. (2001a) 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, and this 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 reticulata* 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.

(ii) 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 *Bufo calamita* tadpoles. Both *B. calamita* and *R. temporaria* are native species in Britain 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

(iii) 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 *Bufo 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 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 Northern Territory of Australia and found that cane toads significantly depleted the abundance of insects (Coleoptera), which 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* 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 the native species, with which they live sympatrically, mainly *Kaloula picta*, *Rana cancrivora*, *Rana vittigera* and *Polypedates leucomystax*, remain abundant.

(iv) 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 experimental manipulations to examine the competitive effects of the larvae of two introduced anurans, the cane toad *Bufo 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.

(v) Coqui *Eleutherodactylus coqui*

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

(vi) Piping frog *Eleutherodactylus johnstonei*

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

(vii) African clawed frog *Xenopus laevis*

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

Reptiles

Based on studies of introduced exotics on Pacific islands and manipulative experiments, Case and Bolger (1991a, b) 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).

(i) 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 pond 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 may exist between introduced *T. scripta* and the native pond turtle *Actinemys marmorata* for basking sites. 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 *Mauremys caspica* in France (Lever 2003), native *Malaclemys caspica* in Israel (Bouskila 1986), and with native *Pelomedusa subrufa* in South Africa (Newberry 1984).

(ii) Common house gecko *Hemidactylus frenatus*

A native, unisexual gecko *Lepidodactylus lugubris* declines numerically when the sexual gecko *Hemidactylus 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) conclude that clumped resources (for example, 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. (2000) who found that *L. lugubris* avoided sharing hiding places with *H. frenatus*.

The introduced house gecko *H. frenatus* was first found in Hawaii in 1951, has competitively displaced the mourning gecko *Lepidodactylus 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 species were introduced to Hawaii by Polynesian travellers about 400 AD and became scarce 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 these habitats on islands where it 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*. Additionally *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*, suggesting *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 *H. frenatus* has caused the catastrophic decline and extinction of the endemic night gecko *Nactus* populations. Neither habitat destruction nor any other introduced competitor or predator can account for the fragmentation of the night geckos 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, thus increasing the risk of exposure 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 former frequently observed stalking, lunging towards and biting the latter. 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 (pers. comm. School Biological Sciences, 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*.

(iii) Italian wall lizard *Podarcis sicula*

The lacertid lizard, *Podarcis 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 (Mediterranean) has reduced the range and eradicated many populations of the native wall lizard *P. raffonei* partly through competitive exclusion.

(iv) Anoles *Anolis* spp

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

The introduced brown anole *Anolis 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 where dense shrub cover exists the two species should be able to co-exist.

Fitch et al. (1989) conducted a field study of *Anolis 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 slow their rate of spread, but the two species can co-exist.

(v) Eastern grass skink *Lampropholis delicata*

West (1979) suggested that introduced *Lampropholis delicata* in New Zealand might compete for food with 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 but 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.

2.5.2 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).
- Rates and patterns of prey consumptions as determined by spatial and age distributions of predator populations.
- Capacity of individual predators to increase prey consumption with increasing prey density (functional response) or for 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 tell us little about the potential significance of exotic species as predators of native fauna. This is because one species predation on another species may not mean it reduces 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 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).

Anurans

Adult anurans 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, hatchling 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).

(i) American bullfrog *Rana catesbeiana*

Originally native to eastern North America, *R. catesbeiana* has been widely introduced in the western United States. In the western states, the bullfrog's enemies (basses, pikes, snapping turtles and water snakes) are absent and *R. catesbeiana* attains high population densities (e.g., Rosen and Schwalbe 1995). Although *R. catesbeiana* tadpoles are strongly herbivorous, mainly eating detritus and algae, in the laboratory *R. catesbeiana* 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 these native 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 are depleted. This 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, whereas 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* on the study sites.

Kiesecker and Blaustein (1997b) 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 (coexisted with *R. catesbeiana*) or allotopic (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.

(ii) Coqui *Eleutherodactylus coqui*

E. coqui is introduced to Hawaii and attains extremely high densities. Natural populations of *E. coqui* can reach densities of 20,570 adults per hectare (Stewart and Rand 1991). Kraus et al. (1999) estimated that in Hawaii *E. coqui* has been able to reach densities ten times greater than has been found in its natural environment in Puerto Rico. It can invade mid-elevation moist and rain forests where it can be expected to exert tremendous predation pressure on a variety of native arthropods (Kraus et al. 1999). Tummons (2003) suggested dense populations of coqui frogs may eat over 200 kilograms of arthropods per hectare per year.

(iii) 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, planigales *Planigale maculata*, 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 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 *B. terrestris* overlap, *B. marinus* preys on *B. terrestris,* and Rossi (1981) suggested such predation may contribute to scarcity of *B. terrestris* 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.

(iv) Cuban treefrog *Osteopilus septentrionalis*

The Cuban tree frog, *O. septentrionalis*, preys on native *Bufo terrestris*, *Gastrophryne carolinensis*, *Rana sphenocephala*, *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 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* reduce populations of native frogs in natural areas.

(v) African clawed frog *Xenopus laevis*

X. laevis 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 and amphibians and terrestrial vertebrates) have also been found in the diet (Lafferty and Page 1997; Measey 1998a, b; Lobos and Measey 2002; Lobos and Jaksic 2005).

In brackish streams and estuaries in California, *X. laevis* were found to have native tidewater gobies *Eucyclogobius newberryi* in their stomachs (Lafferty and Page 1997). Tidewater gobies have declined but other factors are also likely to have contributed.

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 this in Chile.

According to Lafferty and Page (1997) *X. laevis* can prey on vulnerable finfish and could threaten the survival of the tidewater goby *Eucyclogobius 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).

Reptiles

(i) Brown treesnake *Boiga irregularis*

The arrival and proliferation of brown tree snakes *Boiga irregularis* 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 meant roosting and nesting birds, eggs and nestlings were all vulnerable to predation. Following introduction, 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) suggests 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 consume annually 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 ha forested study site.

(ii) Common house gecko *Hemidactylus frenatus*

In the laboratory house geckos prey on juvenile mourning geckos 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) suggests that on Christmas Island, the introduced *H. frenatus* has the potential to affect adversely the endemic Christmas Island gecko *Lepidodactylus listeri.*

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

(iii) Brown anoles *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 *A. carolinensis* in Florida and native *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 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*, and were 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 *A. sagrei* under field conditions. Campbell (1999/2000) suggested 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 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 *A. sagrei* and found that *A. sagrei* 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 the island introduction experiment conducted by Schoener and Spiller (1996): exclusion of rare spider species. In the island introduction experiment, 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. 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) suggest that in their mainland lizard exclosure experiments, spiders were being reintroduced from outside the exclosures but this happened less on isolated islands. Also there may have been fewer refugia for spiders to escape lizard predation on island habitats.

(iv) Crested anole *Anolis cristatellus*

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

(v) 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 (Guam) coincided with decline in populations of Pacific blue-tailed skink *Emoia caeruleocauda* and possible eradication of Marianas blue-tailed skink *Emoia atrocostata* and mottled snake-eyed skink *Cryptoblepharus poecilopleurus* in the following decades (Rodda et al. 1991).

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

According to Martínez-Morales and Cuarón (1999) the boa *Boa constrictor* was introduced onto Cozumel Island, Mexico, and is now widespread, and poses a threat to the existence of endemic and other native terrestrial vertebrates of the island. According to Lever (2003) an anecdotal historical record suggests that the introduced *Boa constrictor* on Cozumel Island in Mexico '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 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. 2004a, 2004b).

Fritts (1993) suggested 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 islands lizards, including Bojer's skink *Gonygylomorphus bogerii*.

Kraus and Cravalho (2001) suggested that several exotic snake species that have been 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 which would increase the risk to native prey species. Several authors also suggest there are hundreds of other snake species worldwide which could have similar devastating effects to *Boiga 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.

2.5.3 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 1998a, b). According to Measey (1998b), predation by aquatic predatory amphibians such as African clawed frog *Xenopus laevis* has the potential to have major impacts on freshwater ecology, particularly when this species is present in high densities. *Xenopus laevis* is a generalist predator that consumes a wide variety and size of invertebrate prey. Lobos and Measey (2002) found exotic *Xenopus 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 *Xenopus laevis* disturbing sediments and a change in population dynamics of native predators.

Kraus et al. (1999) speculated on the potential impacts of the *Eleutherodactylus* (*E. coqu*i 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 *Anolis 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 *Anolis 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 years study. Lizard introduction directly and rapidly reduced spider density to that on the natural lizard islands.

In the Bahamas, introduced *Anolis 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* (*E. coqu*i and *E. planirostris*) frogs in Hawaii could provide an abundant food source for introduced predators, such as rats, cats and mongooses (*Herpestes javanicus*) leading to an increase in their 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 *Boiga irregularis* should they be introduced to Hawaii. Similarly, Zug et al. (1975) suggested an indirect impact that *B. marinus* may have on the Hawaiian ecosystem is that mongoose and rats can prey on *B. marinus*, creating another food source for these invasive predators

According to Fritts and Rodda (1998) introduced *Anolis carolinensis* in Guam provide prey for the introduced brown tree snake *Boiga 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 consequent increased threat to native birds (Fritts & Rodda 1998; Rodda et al. 1999).

According to Wilson and Porras (1983), the population of the exotic corn snake *Elaphe guttata* has grown in urbanized areas of southern Florida as a direct result of the increase in the numbers of exotic *Anolis sagrei.*

(iii) Provide prey for native predators

Exotic reptiles and amphibians may 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* have increased in some urban areas of Florida because increasing numbers of exotic *Anolis sagrei* are available as prey.

In North America the introduced brown anole *Anolis sagrei* is both predator and prey for native species and also a competitor (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 a predator (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 brown anoles *A. sagrei* 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 *Xenopus laevis* in Chile disturb sediments and increase water turbidity and this could have secondary impacts on other biota.

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

On 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 and said this was consistent with an absence of vertebrate seed dispersers due to predation by *Boiga irregularis*.

(v) Indirectly facilitate survival of other exotic species

According to Adams et al. (2003), positive interactions among non-native species can exacerbate the problem of invasions, but are 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 predatory macroinvertebrate densities. Native dragonfly nymphs in Oregon, 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. This 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 *Bufo marinus* on populations of native anuran 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 *L. ornatus* and *L. rubella*. *L. ornatus* tadpoles are voracious predators of *L. rubella* eggs and hatchlings. Consequently, the negative impact of *B. marinus* on populations of *L. ornatus* tadpoles reduced the intensity of predation by *L. ornatus* tadpoles on *L. rubella* eggs and hatchlings, thereby increasing *L. rubella* survival. Crossland's (2000) results demonstrate that *B. marinus* plays an important role in re-structuring native larval anuran communities via direct and indirect mechanisms, and that *B. marinus* may have both negative and positive effects on populations of native anuran larvae.

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

2.5.4 Potential to cause injuries

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

(i) Venomous or toxic bite For example, venomous snakes and some lizards whose bite can lead to blood poisoning.

The brown tree snake *B. irregularis* was accidentally transported from its native range in the South Pacific to Guam. The snakes caused widespread loss of domestic birds and pets and considerable emotional trauma to residents and visitors alike when snakes invaded human habitats with the potential for dangerous venomous bites to small children (Fritts and Leasman-Tanner 2001) although there have been no human fatalities according to Fritts et al. (1994).

(ii) Poisonous skin glands

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

The cane toad *Bufo marinus* is well-protected at all life stages by skin glands that secrete a highly toxic fluid and 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, can be killed by cane toad toxin (Freeland 1984; Lever 2003). Australian native animals that have died due to the ingestion of *B. marinus* include the Australian native cat *Dasyurus geoffroii*, numerous snake species, crows, kookaburras, and the Tasmanian devil *Sarcophilus harrisii* (Covacevich and Archer 1975). Cane toads in Australia are also thought to affect native amphibian populations and other aquatic fauna, 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 (e.g. firetail gudgeon *Hypseleotris galli*) avoid eating cane toad tadpoles and die if they do eat them (Freeland 1984). 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 at the population level, or to refute them (Freeland and Martin 1985). However, many native predators in Australia and elsewhere are susceptible to cane toad toxin. Varanid 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). 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 colonizing cane toads in northern Queensland caused severe population declines in five predator species: *Dasyurus hallucatus, Varanus gouldii, V. mertensi, V. panoptes*, and *V. timorensis similis*. Phillips et al. (2003) predict predation by cane toads has the potential to have significant impacts on some Australian snakes and suggest that cane toads threaten populations of approximately 30% of terrestrial Australian snake species. Crossland (2000) found, under 'naturalistic' conditions, populations of predatory native tadpoles *Limnodynastes ornatus* tadpoles experienced significantly reduced survival when exposed to *B. marinus* eggs and hatchlings.

Animals may learn or evolve traits to avoid cane toad poisoning and so their effects may be temporary. For example, Crossland (2001) found 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 recognized and avoided tadpoles one day after trial encounters and no fish died in the trials. Phillips and Shine (2004) found two species of native snakes in Australia, *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 more time. In Florida, *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 cat numbers declined when *B. marinus* first arrived in Dumaguete City in the Philippines, due to cane toad poisoning, but cats learnt to avoid cane toads and cat numbers recovered (Rabor 1952; Alcala 1957). Crossland and Azevedo-Ramos (1999) offered dead *B. marinus* tadpoles as food to tadpoles of native species from Brazil and Australia. The native tadpoles from Brazil ate the dead *B. marinus* without apparent ill effects whereas the majority of the native tadpoles from Australia died after eating the *B. marinus* tadpoles. Apparently the tadpoles from Brazil, which had co-evolved with *B. marinus,* had developed resistance to cane toad toxins.

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 affected negatively. Dogs are known to die within 15 minutes of mouthing a cane toad.

(iii) Organs and or body size capable of causing physical injury For example, crocodiles may be over six metres long and may weigh close to 1000 kilograms and have strong jaws capable of crushing and teeth capable of tearing flesh.

(iv) Traffic hazard In Australia *B. marinus* are considered 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.

2.5.5 Role as disease carriers and reservoirs

Diseases spread by exotic reptiles and amphibians to native species may have ecological consequences. Exotic reptiles and amphibians can serve as hosts, reservoirs and vectors for diseases and parasites that affect human and animal health. Examples include:

- Red-eared slider *Trachemys scripta*

The United States Food and Drug Administration has banned the sale of turtles under four inches because they can transmit the disease salmonellosis which can be transferred to humans via drinking water (Newberry 1984; USGS 2003a). Once enormously popular in the United States as pets, millions of red-eared sliders *T. scripta* 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* in Washington were decimated by a respiratory infection in 1990 and Hays et al. (1999) implicate introduced *T. scripta* as a likely vector for the infection. Spinks et al. (2003) found a female *T. scripta* in a California waterway showed 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 *Bufo marinus*

According to Freeland (1984) cane toads in Australia eat human faeces and may thus spread parasites such as *Trichuris trichiura* and *Schistoma mansoni*, spread eggs of canine hookworm *Uncinaria* and possibly human hookworm (unspecified species), and spread *Salmonella.* 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 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 canetoads (Speare 1990). Whether diseases or parasites carried by cane toads have negative effects on native species has not been investigated (Boland 2004).

On St Lucia in the West Indies, *B. marinus* are claimed to harbour ticks that affect cattle Lever (2003).

- African clawed frog *Xenopus 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.

2.5.6 Hybridisation with native species and other genetic changes

When exotic reptiles and amphibians hybridise with native reptiles and amphibians, and produce fertile offspring, this corrupts the gene pool of the native reptiles and amphibians and hence may pose a threat to their survival.

Changes in the genetic structure of a population can occur due to reductions in size, reduced numbers of subpopulations or phenotypes, due to competition, habitat alterations or predations (Elvira 2001).

Even a few 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) suggest risks are highest when rare species hybridize with an abundant species and the offspring are fertile and can back-cross (introgression). Even without introgression, hybridization 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 the pressures exerted through introductions that can change normal behaviour patterns, hybrids arise between species or genera that would not otherwise interbreed (Elvira 2001).

Butterfield et al. (1997) consider hybridization 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. (2002) and Riley et al. (2003) examined hybridization 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 ponds sampled. These hybrids were viable and fertile. Despite a relatively ancient split and wide genetic divergence between these taxa, they are interbreeding and threatening the genetic purity of the native species. Four artificial ponds had greater genetic mixing than two natural ponds.

Other possible examples of hybridization in exotic reptiles and amphibians include:

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

Exotic species can have genetic effects other than hybridisation. They may have indirect genetic effects by altering patterns of natural selection or gene flow in native species 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 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.

2.5.7 Social and economic impacts

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

Coqui *Eleutherodactylus coqui*

E. coqui is a tiny frog from Puerto Rico with loud, piercing calls that can measure 90–100 decibels at a distance of 0.5 metres from a frog. The frog can reach high densities where it has been introduced in Hawaii and the 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). Residents are encountering reduced property values and increased difficulty selling property (Kraus and Campbell 2002). This is also a problem for other areas where *Eleutherodactylus* species have been introduced outside their native ranges. 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 has recently initiated a ban on nursery products coming from Hawaii because of receipt of E. coqui in such shipments (E. Campbell, pers. comm., US Fish and Wildlife Service, Honolulu, 2005]. Another negative consequence of the spread of *Eleutherodactylus coqui* in Hawaii is the illegal use of toxic chemicals by residents attempting to kill the frogs (Kraus and Campbell 2002).

Brown tree snake *Boiga irregularis*

Due to its arboreal nature, *B. irregularis* snakes climbing on electrical lines have become a huge economic burden in Guam. The snakes short out electrical systems and cause extensive electrical damage, affecting private, commercial, and military activities, causing 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

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

Cane toad *Bufo marinus*

Introduced *B. marinus* pollute freshwater with eggs and tadpoles in Japan (Lever 2003). Similarly in Bermuda, where the sole source of fresh drinking water is rainwater tanks, introduced *B. marinus* enter and drown in these tanks and pollute the water. In Australia *B. marinus* pollute and block water supply, drainage and storage facilities including swimming pools; their decomposing bodies pollute other water bodies; they cause erosion of earth dams and creek banks by burrowing; and 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 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, *B. marinus* is considered a pest in nurseries because it buries itself in potting mix and destroys seedlings and it also damages 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 *Osteopilus septentrionalis*

In the West Indies, introduced *O. septentrionalis* are a problem because they invade drinking water tanks, cisterns and toilet vent pipes (Lever 2003).

African clawed frog *Xenopus laevis*

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

Potential damage to aquaculture facilities

Although no reports of harm to aquaculture facilities caused by exotic amphibians were found, the potential for such harm does exist. For example, *Rana catesbeiana* 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 frogs showed fish (both species) were the highest volume of food eaten by bullfrogs. In the goldfish ponds bullfrogs 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.

2.6 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 which 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 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 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 may not have been introduced outside their natural range yet, 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 to 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 predict the pest potential of future reptiles and amphibians 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 (bullfrog *Rana catesbeiana*; cane toad *Bufo marinus*; Caribbean tree frog or coqui *Eleutherodactylus coqui*) and two reptile (brown tree snake *Boiga irregularis*; red-eared slider *Trachemys scripta*) species.

Taxa that have novel adaptations 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 (see Section 2.6 iv). For example, all species in the genera *Anolis* and *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-aggregated 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:

- Around 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 *Eleutherodactylus coqui* in its native forests in Puerto Rico (Stewart and Rand 1991; Beard et al. 2003) and even higher densities in its introduced range in Hawaii (Kraus 2000) including one estimate of 37 000 per hectare (Tummons 2003).
- Nearly 30 000 per hectare for North American red-backed salamander *Plethodon cinereus* (Campbell and Echternacht 2003).

- 23 600 per hectare for anole *Anolis stratulus* (Reagan 1992).
- At least 12 000 per hectare for brown anoles *Anolis sagrei* introduced in Florida (Campbell and Echternacht 2003)
- At least 3700 per hectare for *Xenopus laevis* in its introduced range in Chile (Lobos and Measey 2002).
- 46.3 per hectare (12 000 snakes per square mile) for the brown tree snake *Boiga 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 the native or introduced ranges should be considered to pose a high risk of impact in 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.

2.7 Discussion

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. Of the hundreds of exotic reptiles and amphibians introduced around the world, only six (*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. One explanation for this lack of attention may be that exotic reptiles and amphibians are not often economic or agricultural pests and they are also not often viewed as an ecological threat. Their ecological impacts on native species and communities are not usually obvious to people not trained in ecology, especially in comparison to such species as large predatory mammals.

The impacts of exotic reptiles and amphibians are most readily recognized when an abundant introduced species leads to major declines in native species, for example the brown treesnake *Boiga 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 and 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, including competition and hybridisation with, and predation on native species, disruption of ecosystem trophic dynamics, and negative economic and social impacts (Kraus and Campbell 2002; Bomford 2003; Bomford and Glover 2004).

There is insufficient reliable knowledge of the factors correlated with impacts of exotic reptiles and amphibians to make the development of a quantitative model feasible for assessing the risks of impact for new species of exotic reptiles and amphibians in Australia. Nonetheless, the review of factors associated with adverse impacts presented in this section 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. This would be particularly desirable if decisions are being made on whether to import species that score a moderate or higher risk of establishment in the quantitative risk assessment model developed in Section 3.

Section 3: Simple model to discriminate between exotic reptile and amphibian species successfully or unsuccessfully introduced to Australia

3.1 Aim

To develop a simple quantitative model to predict risk of establishment of exotic reptiles and amphibians introduced to Australia based on an analysis of exotic reptile and amphibian introductions to Britain, California and Florida.

Section 1.2.1 of this report demonstrates that there are four key factors associated with the establishment success of exotic reptiles and amphibians in Australia:

(i) Number of release events
(ii) Climate match
(iii) History of establishing exotic populations elsewhere
(iv) Taxonomic group

Factor (i) means that the risk of new exotic reptile and amphibian species establishing in Australia can be expected to increase as the number of people keeping exotic reptiles and amphibians increases and the numbers of different reptile and amphibian species kept in collections increases. This is because, as more people keep reptiles and amphibians, the number of escapes and releases of new reptile and amphibian species will also increase, and establishment of exotic reptiles and amphibians is closely correlated with the number of release events (Figure 1). This risk can be reduced by restricting the import and keeping of reptiles and amphibians species which are ranked highly against the other three factors. The number of release events is not included as a factor in the risk model.

3.2 The model

The model developed in this section provides a simple quantitative method for ranking reptiles and amphibians against these three factors:

- Score A: CLIMATE match
- Score B: History of establishing exotic populations elsewhere
- Score C: Taxonomic family

The sum of these three scores = a species' Establishment Risk Score.

The threshold values presented for calculating risk scores A, B and C below were all selected to give the most accurate predictive outcome (success or failure to establish an exotic population) for exotic reptiles and amphibians introduced to Britain, California and Florida. An assumption is made that exotic reptiles and amphibians introduced to Australia will follow the same general pattern as species introduced to these other regions and so a matching risk score will equate to a matching risk of an exotic species establishing in Australia.

Score A: Climate Match Risk Score

CLIMATE uses temperature and rainfall data from a set of geographical locations to construct a climate profile. This is used to indicate geographical regions which are contained within the boundaries of the profile. Typically, the profile is based on the locations where an entity, such as a reptile species, is known to occur overseas. The program then matches the constructed profile to Australia.

Use CLIMATE (Version 2 for PCs), with '*worlddata_all.txt*' selected as the world data location and select the shapefile '*Merge.shp*' (which has US states outlined), or 'cntry92.shp', (which does not have US states outlined), (Knapp et al. in prep. 2005). Select all 16 available climatic parameters for matching locations (Table 3). Select 'Euclidian match' for the analysis.

Table 3. Parameters used in climate matching analysis using CLIMATE Version 2.

Parameter	**Parameter description**
Avrf	Average annual rainfall
Avt	mean annual temperature
Coefvar	coefficient of variation of monthly rainfall
Coolmth	Minimum temperature of coolest month
Coolq	mean temperature of coolest quarter
Drymth	Rainfall of driest month
Dryq	Rainfall of driest quarter
Rfcoolq	Rainfall of coolest quarter
Rfhotq	Rainfall of warmest quarter
Spredt	Average temperature range
Tdryq	mean temperature of driest quarter
Twetq	mean temperature of wettest quarter
Warmth	max temp of warmest month
Warmq	mean temp of warmest quarter
Wetmth	Rainfall of wettest month
Wetq	Rainfall of wettest quarter

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

For example, the cane toad gets match scores to Australia of:
Number 7 match = 857
Number 8 match = 951
Number 9 match = 41
Number 10 match = 0
Σ 7–10 matches = 1849
Score A = Climate match score = 100×(1849÷2785) = 66%

Score B: Exotic Elsewhere Risk Score

Score B = A species' Exotic Elsewhere Risk Score =

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

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

Score C: Taxonomic family risk score

Score C = A species' Taxonomic Family Risk Score is taken from Table 4.

- 30 = Extreme risk
- 20 = Very high risk
- 15 = High risk
- 10 = Moderate risk
- 5 = Low risk
- 0 = Very low risk

For example, the cane toad is in Family Bufonidae and gets a Very High Taxonomic Family Risk Score = 20.

Table 4. Taxonomic Family Risk Scores for exotic reptiles and amphibians (Based on data sourced from F. Kraus, unpublished database – see Section 1.2.1 iv).

Family	**Successful introduction events %**	**Taxonomic Family Risk Score**
Dendrobatidae	100	30
Proteidae	100	30
Typhlopidae	95	30
Ranidae	80	30
Leptodactylidae	79	30
Chamaeleonidae	79	30
Gekkonidae	76	30
Rhacophoridae	75	30
Agamidae	70	30
Teiidae	67	20
Trionychidae	66	20
Bufonidae	60	20
Microhylidae	60	20
Plethodontidae	58	20
Lacertidae	57	20
Iguanidae	56	20
Testudinidae	48	15

Scincidae	46	15
Pipidae	42	15
Hylidae	41	15
Myobatrachidae	40	15
Emydidae	39	15
Discoglossidae	38	15
Ambystomatidae	38	15
Varanidae	38	15
Salamandridae	36	15
Anguidae	29	10
Chelydridae	29	10
Pelomedusidae	25	10
Chelidae	22	10
Viperidae	21	10
Colubridae	20	10
Cordylidae	17	10
Alligatoridae	15	10
Elapidae	11	10
Boidae	6	5
Pelobatidae	0	0
Cryptobranchidae	0	0
Amphisbaenidae	0	0
Gymnophthalmidae	0	0
Helodermatidae	0	0
Pygopodidae	0	0
Kinosternidae	0	0
Crocodylidae	0	0
Geomydidae	0	0

Establishment Risk Score

A species’ Establishment Risk Score = Score A + Score B + Score C. Establishment Risk Scores can be converted to Establishment Risk Ranks ranging from Very Low to Extreme using the conversions levels presented in Table 5.

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

Table 5. Conversion of Establishment Risk Scores to Establishment Risk Ranks.

Establishment Risk Rank	**Establishment Risk Score range**
Very Low	< 20
Low	20–45
Moderate	46–60
High	61–84
Very High	85–115
Extreme	≥ 115

Figure 5 presents the combined establishment risk rankings for all the exotic reptile and amphibian species introduced to Britain, California and Florida. This shows clearly that most species that failed to establish have very low or low establishment risk ranks, whereas most successful species have a moderate or higher ranking.

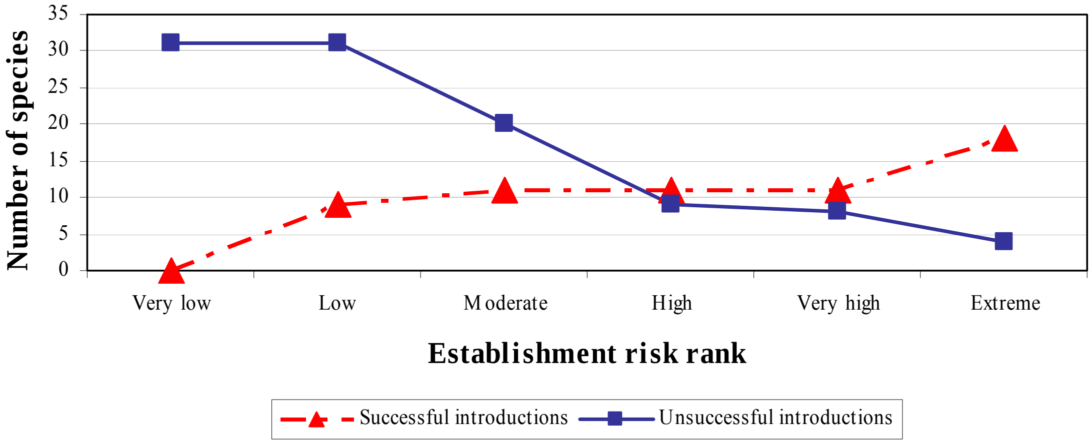

Figure 5. Combined Establishment Risk Ranks for reptiles and amphibians introduced to Britain, Florida and California. 85% (51/60) of successfully introduced species have Moderate or higher risk ranks. 80% (82/103) of failed species have Moderate or lower risk ranks. Source data in Appendix A.

3.3 Discussion

The model presented in this section for predicting establishment risk for exotic reptiles and amphibians introduced in Australia is quantitative, repeatable and transparent, and does not rely on subjective judgement.

The Establishment Risk Rankings presented in Figure 5 show a good correlation between establishment success and Establishment Risk Rank. But not all successfully established species have a high Establishment Risk Rank and not all failed species have a low Establishment Risk Rank. One possible reason for this is that the establishment risk ranks do not take account of propagule pressure – the number of individual animals released and the number of times release events occur for each species. Propagule pressure has a strong influence on introduction outcomes (see Section 1.2.1 i), so this factor probably accounts for some of the unexplained variance in Figure 5. This means that some of the unsuccessful species that have a high Establishment Risk Rank, might well establish exotic populations if further releases of these species occur.

Acknowledgements

This project was conducted for the Department of the Environment and Heritage and funded by the Natural Heritage Trust.

References

Adams M.J.; Pearl C.A.; Bruce Bury R. 2003. Indirect facilitation of an anuran invasion by non-native fishes *Ecology Letters* 6: 343–351.

Adolph, S.C. and Roughgarden, J. 1983. Foraging by passerine birds and *Anolis* lizards on St Eustatius (Neth. Antilles): implications for interclass competition and predation. *Oecologia* 56: 313–317.

Alcala, A.C. 1957. Philippine notes on the ecology of the giant marine toad. *Silliman Journal* 4: 90–96.

Alcover, J.A., and Mayol, J. 1981. Espècies relíquies d'amfibis i de rèptils a les Balears i Pitüses. *Bolletí de la Societat d'Història Natural de les Balears* 25: 151–167.

Amand, A. 2000. *Boiga irregularis* (Brown Tree Snakes) on Guam and its effect on fauna. *Restoration and Reclamation Review* 6: Fall.

Aquatic Nuisance Species Taskforce. 1996. Report to the Aquatic Nuisance Species Task Force – Generic Nonindigenous Aquatic Organisms Risk Analysis Review Process. http://www.anstaskforce.gov/gennasrev.htm

Arthington, A.H., Kailola, P.J. Woodland, D.J. and Zalucki, J.M. 1999. Baseline environmental data relevant to an evaluation of quarantine risk potentially associated with the importation to Australia of ornamental finfish. Report to the Australian Quarantine and Inspection Service, Department of Agriculture, Fisheries and Forestry, Canberra.

Ashton, R.E. Jr. and Ashton, P.S. 1988. *Handbook of reptiles and amphibians of Florida*. Windward Publishing Inc, Miami, Florida.

Babbitt, K. and Meshaka, W. 2000. Benefits of eating conspecifics: effects of background diet on survival and metamorphosis in the Cuban treefrog (*Osteopilus septentrionalis*). *Copeia* 2000: 469–474.

Bartlett, R.D. and Bartlett, P.B. 1999. *A field guide to the Florida reptiles and amphibians*. Gulf Publishing Company, Houston, Texas.

Beard, K.H., Eschtruth, K., Vogt, K.A. Vogt, D.J. and Scatena, F.N. 2003. The effects of the frog *Eleutherodactylus coqui* on invertebrates and ecosystem processes at two scales in the Luquillo Experimental Forest, Puerto Rico. *Journal of Tropical Ecology* 19: 607–617.

Beebee, T.J.C. 1995. http://www.open.ac.uk/daptf/froglog/FROGLOG_12-1.html

Beller, D. 1997. Evaluating the Potential Role of Bullfrogs in Aquatic Degradation in Southern Nevada. Bachelor of Arts thesis, Environmental Studies Program, University of Nevada, Las Vegas. http://environment.unlv.edu/thesis/aug1997/beller.htm

Boland, C.R.J. 2004. Introduced cane toads *Bufo marinus* are active nest predators and competitors of rainbow bee-eaters *Merops ornatus*: observational and experimental evidence. *Biological Conservation* 120: 53–62.

Bomford, M. 2003. *Risk Assessment for the Import and Keeping of Exotic Vertebrates in Australia.* Bureau of Rural Sciences, Canberra.

Bomford, M. and Glover, J. 2004. Risk assessment model for the import and keeping of exotic freshwater and estuarine finfish. Bureau of Rural Sciences, Canberra.

Bouskila, A. 1986. On the danger of red-eared terrapin, *Chrysemys scripta*, in natural habitats in Israel. *Hardun* 3: 63.

Boyd, S.H. 1975. Inhibition of fish reproduction by *Rana catesbeiana* larvae. *Physiological Zoology* 48: 225–234.

Brown, J.H. 1989. Patterns, modes and extents of invasions by vertebrates. Pp. 85–109 in: Drake, J.A., Mooney, H.A., di Castri, F., Groves, R.H., Kruger, F.J., Rejmanek, M. and Williamson, M.W. (eds) *Biological Invasions. A Global Perspective*. John Wiley and Sons, Chichester.

Brown, S.G., Lebrun, R., Yamasaki, J., Ishii-Thoene, D., 2002. Indirect competition between a resident unisexual and an invading bisexual gecko. *Behaviour* 139: 1161–1173.

Burnett, S. 1997. Colonising cane toads cause population declines in native predators: reliable anecdotal information and management implications. *Pacific Conservation Biology* 3: 65–72.

Bury, R.B. and Luckenbach, R.A. 1976. Introduced reptiles and amphibians in California. *Biological Conservation* 10: 1–14.

Bury, R.B. and J.A. Whelan. 1984. Ecology and management of the bullfrog. U.S. Dept. of the Interior, Fish and Wildlife Service Resource Publication 155.

Butterfield, B.P., Meshaka, W.E. Jr., and Guyer, C. 1997. Nonindigenous amphibians and reptiles. Pp 123–137 in: Simberloff, D., Schmitz, D.C., and Brown, T.C. (eds). *Strangers in Paradise: Impact and Management of Non-indigenous Species in Florida*. Island Press, Washington DC.

Byers, J.E., Reichard, S., Randall, J.M. et al. 2002. Directing research to reduce impacts of nonindigenous species. *Conservation Biology* 16: 630–640.

Cadi, A., Delmas, V., Prévot-Julliard, A.C., Joly, P., Pieau, C. and Girondot, M. 2004. Successful reproduction of the introduced slider turtle (*Trachemys scripta elegans*) in the South of France. *Aquatic Conservation: Marine and Freshwater Ecosystems* 14: 237– 246.

Cadi, A. and Joly, P. 2003. Competition for basking places between the endangered European pond turtle (*Emys orbicularis galloitalica*) and the introduced red-eared slider (*Trachemys scripta elegans*). *Canadian Journal of Zoology* 81 1392–1398.

Cadi, A. and Joly, P. 2004. Impact of the introduction of the red-eared slider (*Trachemys scripta elegans*) on survival rates of the European pond turtle (*Emys orbicularis*). *Biodiversity and Conservation* 13: 2511–2518.

Campbell, E.W. III. 1996. The effect of brown tree snake (*Boiga irregularis*) predation on the island of Guam's extant lizard assemblages. Unpublished PhD thesis, Ohio State University, Columbus, Ohio, USA

Campbell, T.S. 1999/2000. The Cuban brown anole invasion in Florida: it's not easy being green. *Aliens* 10: 4–5.

Campbell, T.S. 2000. Analyses of the effects of an exotic lizard (*Anolis sagrei*) on a native lizard (*Anolis carolinensis*) in Florida, using islands as experimental units. PhD Thesis, Department of Ecology and Evolutionary Biology, University of Tennessee, Knoxville, Tennessee.

Campbell, T.S. and Echternacht, A.C. 2003. Introduced species as moving targets: changes in body sizes of introduced lizards following experimental introductions and historical invasions. *Biological Invasions* 5: 193–212.

Capula, M. 1993. Natural hybridization in *Podarcis sicula* and *P. wagleriana* (Reptilia: Lacertidae) *Biochemical Systematics and Ecology* 21: 373–380.

Capula, M. 1994. Genetic variation and differentiation in the lizard, *Podarcis wagleriana* (Reptilia: Lacertidae). *Biological Journal of the Linnean Society* 52: 177–196.

Capula, M., Luiselli, L., Bologna, M.A. and Ceccarelli, A. 2002. The decline of the Aeolian wall lizard, *Podarcis raffonei*: causes and conservation proposals. *Oryx* 36: 66–72.

Case, T.J. and Bolger, D.T. 1991a. The role of introduced species in shaping the abundance and distribution of island reptiles. *Evolutionary Ecology* 5: 272–290.

Case, T.J. and Bolger, D.T. 1991b. The role of interspecific competition in the biogeography of island lizards. *Trends in Ecology and Evolution* 6: 135–139.

Case, T.J., Bolger, D.T. and Petren, K. 1994. Invasions and competitive displacement among house geckos in the tropical Pacific. *Ecology* 75: 464–477

Catling, P.C., Hertog, A., Burt, R.J., Wombey, J.C. and Forrester, R.I. 1999. The short-term effect of cane toads (*Bufo marinus*) on native fauna in the Gulf Country of the Northern Territory. *Wildlife Research* 26: 161–185.

Clarke, G., Grosse, S., Matthews, M., Catling, P., Baker, B., Hewitt, C., Crowther, D. and Sadlier, S. 2001. Environmental Pests in Australia. CSIRO, Canberra. (Available on request from geoff.clarke@csiro.au).

Cogger, H.G., Sadlier, R.A. and Cameron, E.E. 1983. The terrestrial reptiles of Australia's island territories. *Australian National Parks and Wildlife Service Special Publication* 11: 1–80.

Cole, N. 2002. The decline of Mauritian night and day geckos. http://www.bio.bris.ac.uk/research/mammal/gecko.html

Cole, N.C., Jones, C.G. and Harris, S. 2005. The need for enemy-free space: the impact of an invasive gecko on island endemics. *Biological Conservation*125: 467–474.]

Corn, P. S. 1994. What we know and don't know about amphibian declines in the west. Pages 59–67 in W. W. Covington and F. D. Leonard, editors. Sustainable ecological systems: implementing an ecological approach to land management. U.S. Forest Service, Rocky Mountain Forest and Range Experimental Station, Fort Collins, Colorado, USA.

Covacevich, J. and Archer, M. 1975. The distribution of the cane toad, *Bufo marinus*, in Australia and its effect on indigenous vertebrates. *Memoirs of the Queensland Museum* 17: 305–310.

Crawley, M.J. 1986. The population biology of invaders. *Philosophical Transactions of the Royal Society of London* B 314: 711–731.

Crawley, M. J. 1989. What makes a community invasible? *Symposia of the British Ecological Society* 26: 429–53.

Crockett, M.E., Dove. A.D., Rice, K,G., Waddle, J.H. and Percival, H.F. 2002. Effects of the nonindigenous treefrog, *Osteopilus septentrionalis*, on native Hylids in protected areas of South Florida. http://cars.er.usgs.gov/posters/Everglades/Cuban_Treefrog/cuban_treefrog.html

Crossland M.R. 1997. Impact of eggs, hatchlings and tadpoles of the introduced cane toad *Bufo marinus* (Anura: Bufonidae) on native aquatic fauna in northern Queensland, Australia. PhD. Thesis, James Cook University of North Queensland, Townsville.

Crossland, M.R. 1998. A comparison of cane toad and native tadpoles as predators of native anuran eggs, hatchlings and larvae. *Wildlife Research* 25: 373–381.

Crossland, M.R. 2000. Direct and indirect effects of the introduced toad Bufo marinus (Anura: Bufonidae) on populations of native anuran larvae in Australia. *Ecography* 23: 283–290.

Crossland M.R. 2001. Ability of predatory native Australian fishes to learn to avoid toxic larvae of the introduced toad *Bufo marinus.* *Journal of Fish Biology* 59: 319–329.

Crossland, M.R. and Alford, R.A. 1998. Cane toad. *Australian Journal of Ecology* 23: 129–137.

Crossland M.R. and Azevedo-Ramos, C. 1999. Effects of *Bufo* (Anura: Bufonidae) toxins on tadpoles from native and exotic *Bufo* habitats. *Herpetologica* 55: 192–199 Da Silva, E. and Blasco, M. 1995. *Trachemys scripta elegans* in southwestern Spain. *Herpetological Review* 26: 133–134.

Daehler, C.C. and Gordon, D.R. 1997. To introduce or not to introduce: tradeoffs of nonindigenous organisms. *Trends in Ecology and Evolution* 12: 424–425.

Daehler, C. C. and Strong, D.R.J. 1993. Prediction and biological invasions. *Trends in Ecology and Evolution* 8: 380.

Davidson, C., Shaffer, H.B., and Jennings, M.R. 2001. Declines of the California red-legged frog: climate, UV-B, habitat, and pesticides hypotheses. *Ecological Applications* 11: 464–479.

Davidson, C., Shaffer,H.B. and Jennings, M.R. 2002. Spatial tests of the pesticide drift, habitat destruction, UV-B, and climate-change hypotheses for California amphibian declines. *Conservation Biology* 16: 1588–1601.

Day, D. 2005. Cane toads threaten Darwin drinking water. *ABC News Online* Saturday, May 21.

Delvinqueir, B.L. J. and Freeland, W.J. 1988. Protozoan parasites of the cane toad, *Bufo marinus*, in Australia. *Australian Journal of Zoology* 36: 301–316.di Castri, F. 1991. The biogeography of Mediterranean animal invasions. Pp. 439–452 in: R.H. Groves and F. di Castri (eds) *Biogeography of Mediterranean Invasions*. Cambridge University Press, Cambridge.

Dial, R. and Roughgarden, J. 1995. Experimental removal of insectivores from rain forest canopy: direct and indirect effects. *Ecology* 76: 1821–1834.

Downes, S., Bauwens, D. 2004. Associations between first encounters and ensuing social relations within dyads of two species of lacertid lizards. *Behavioral Ecology* 15: 938–945.

Duncan, R.P., Bomford, M., Forsyth, D.M. and Conibear, L. 2001. Correlates of introduction success and geographical range size in introduced Australian birds. *Journal of Animal Ecology* 70: 621–632.

Ebenhard, T. 1988. Introduced birds and mammals and their ecological effects. *Swedish Wildlife Research Viltrevy* 13: 1–107.

Ehrlich, P.R. 1986. Which animal will invade? Pp. 79–95 in H.A. Mooney and J.A. Drake (eds) *Ecology of Biological Invasions in North America and Hawaii*. Springer-Verlag, New York.

Ehrlich, P.R. 1989. Attributes of invaders and the invading process: vertebrates. Pp. 315–328 in: J.A. Drake., H.A. Mooney, F. di Castri, R.H. Groves, F.J. Kruger, M. Rejmanek and M.W. Williamson (eds) *Biological Invasions. A Global Perspective*. John Wiley and Sons, Chichester.

Elvira, B. 2001. Identification of non-native freshwater fishes established in Europe and assessment of their potential threats to the biological diversity. Convention on the conservation of European Wildlife and Natural Habitats Standing Committee 21st meeting, Strasbourg.

Enserink, M. 1999. Biological invaders sweep in. *Science* 285: 1834–1836. FAO 1998. http://www.fao.org/fi%2A/statist/fisoft/dias/mainpage.htm

Easteal, S. 1981. The history of introduction of *Bufo marinus* (Amphibia:Anura); a natural experiment in evolution. *Biological Journal of the Linnean Society* 16: 93–113.

Fisher, R.N. and Shaffer, H.B. 1996. The decline of amphibians in California’s great central valley. *Conservation Biology* 10: 1387–1397.

Fitch, H.S., Henderson, R.W. and Guarisco, H. 1989. Aspects of the ecology of an introduced anole: *Anolis cristatellus* in the Dominican Republic. *Amphibia Reptilia* 10: 307–320.

Forsyth, D.M., Duncan, R.P., Bomford, M. and Moore, G. 2004. Climatic suitability, life-history traits, introduction effort and the establishment and spread of introduced mammals in Australia. *Conservation Biology* 18: 557–569.

Foulis, S.L. and King, A. 1999. Typical spiralling whitefly infestation levels on plant hosts in the Cairns region. Draft. DPI Queensland Department of Primary Industries: Cairns

Freeland, W.J. 1984. Cane toads: a review of their biology and impact in Australia. Technical report No 19. Conservation Commission of the Northern Territory, Darwin.

Freeland, W.J. 1987. Cane toads and the balance of nature. *Wildlife Australia* 24: 12–15.

Freeland, W.J., Delvinqueir, B.L. J. and Bonnin, B. 1986. Food and parasitism of the cane toad, *Bufo marinus*, in relation to time since colonization. *Australian Wildlife Research* 13: 489–499.

Freeland, W.J. and Kerin, S.H. 1988. Within-habitat relationships between invading *Bufo marinus* and Australian species of frog during the tropical dry season. *Australian Wildlife Research* 15: 293–305.

Freeland, W.J. and Martin, K. C. 1985. The rate of range expansion by *Bufo marinus* in northern Australia, 1980–84. *Australian Wildlife Research* 12: 555–559.

Fritts, T.H. 1988. The brown tree snake, *Boiga irregularis*, a threat to Pacific islands. *US Fish and Wildlife Service Biological Report 88.* Washington DC.

Fritts, T.H. 1993. The common wolf snake, *Lycodon aulicus capucinus*, a recent colonist of Christmas Island in the Indian Ocean. *Wildlife Research* 20: 261– 266.
Fritts, T.H., and Leasman-Tanner, D. 2001. The brown tree snake on Guam: how the arrival of one invasive species damaged the ecology, commerce, electrical systems, and human health on Guam: A comprehensive information source. Available Online: http://www.fort.usgs.gov/resources/education/bts/bts_home.asp

Fritts, T. H. and McCoid, M.J. 1991. Predation by the brown tree snake *Boiga irregularis* on poultry and other domesticated animals in Guam. Snake 23: 75–80.

Fritts, T.H., McCoid, M.J. and Haddock, R.L. 1994. Symptoms and circumstances associated with bites by the brown tree snake (Colubridae: *Boiga irregularis*) on Guam. *Journal of Herpetology* 28: 27–33.

Fritts, T.H. and Rodda, G.H. 1995. Invasions of the brown tree snake. Pages 454-456 in Laroe, E. T., G. S. Farris, C. E. Puckett, P. D. Doran, and M. J. Mac, editors. Our living resources: a report to the nation on the distribution, abundance, and health of U.S. plants, animals, and ecosystems. U.S. Department of the Interior, National Biological Service, Washington, D.C.

Fritts, T. H. and Rodda, G.H. 1998**.** The role of introduced species in the degradation of island ecosystems: a case history of Guam. *Annual Review of Ecology and Systematics* 29: 113–140.

Fryer, G. 1991. Biological invasions in the tropics: hypothesis versus reality. Pp. 87–101 in: Ramakrishnan, P.S. (ed.) *Ecology of Biological Invasions in the Tropics*. International Scientific Publications, New Delhi.

Gerber, G.P. 1999. A review of intraguild predation and cannibalism in *Anolis*. Pp28–39 in Losos, J.B. and Leal, M. (eds) *Anolis* Newsletter V. Washington University, St Louis.

Gerber, G.P. and Echternacht, A.C. 2000. Evidence for asymmetrical intraguild predation between native and introduced *Anolis* lizards. *Oecologia* 124: 599–607.

Gilpin, M. 1990. Ecological Prediction. *Science* 248: 88–89.

Gorman, G.C., Kim, Y.J. and Yang, S.Y. 1978. The genetics of colonization: loss of variability among introduced populations of *Anolis* lizards (Reptilia, Iguanidae). *Journal of Herpetology* 12: 47–51.

Gorman, G.C. and Atkins, L.1968. Natural hybridization between two sibling species of *Anolis* lizards: chromosome cytology. *Science* 159: 1358–1360.

Gorman, G.C., Licht, P., Dessauer, H.C. and Boos, J.O. 1971. Reproductive failure among the hybridizing *Anolis* lizards of Trinidad. *Systematic Zoology* 20: 1–18.

Guisan, A. and Hofer, U. 2003. Predicting reptile distributions at the mesoscale: relation to climate and topography. *Journal of Biogeography* 30: 1233–1243.

Griffiths, R.A. 1991. Competition between common frog, *Rana temporaria*, and natterjack toad, *Bufo calamita*, tadpoles: the effect of competitor density and interaction level on tadpole development. *Oikos* 61: 187–196.

Hammerson, G.A. 1982. Bullfrog eliminating leopard frogs in Colorado? *Herpetological Review* 13: 115–116.

Hayes, M.P. and Jennings, M.R. 1986. Decline of Ranid frog species in western North America: are bullfrogs (*Rana catesbeiana*) responsible? *Journal of Herpetology* 20: 490–509.

Hayes, K.R. and Sliwa, C. 2003. Identifying potential marine pests – a deductive approach applied to Australia. *Marine Pollution Bulletin* 46: 91–98.

Hayes, K., Sliwa, C., Migus, S., McEnnulty, F. and Dunstan, P. 2004. National priority pests: Part II: Ranking of Australian marine pests. An independent report undertaken for the Department of Environment and Heritage by CSIRO Marine Research. http://www.marine.csiro.au/crimp/reports/PriorityPestsFinalreport.pdf

Hays, D.W., McAllister, K.R., Richardson, S.A. and Stinson, D.W. 1999. Washington State Recovery Plan for the Western Pond Turtle. Washington Department of Fish and Wildlife, Olympia.

Hecnar, S. J., and R. T. M'Closkey. 1997. Changes in the composition of a ranid frog community following bullfrog extinction. *American Midland Naturalist* 137:145–150.

Holmes, B. 1998. Day of the sparrow. *New Scientist* (27 June): 32–35.

Hutchinson, M. 2001. Tealeaves and Tumbling Dice - Predicting Animal Invasions. Proceedings of Exotic Reptile & Amphibian Conference. http://homepages.ihug.com.au/~venomsup/exoticconference/markhutchinsonstalk.html

Kaiser, H., Hardy, J.D.Jr and Green, D.M. 1994. The taxonomic status of Caribbean and South American frogs currently ascribed to *Eleutherodactylus urichi* (Anura: Leptodactylidae). *Copeia* 1994: 780–796.

Kiesecker, J.M. and Blaustein, A.R. 1997b. Population differences in responses of red-legged frogs (*Rana aurora*) to introduced bullfrogs (*Rana catesbeiana*). *Ecology* 78: 1752–1760.

Kiesecker, J. M. and Blaustein, A.R. 1998. Effects of introduced bullfrogs and smallmouth bass on microhabitat use, growth, and survival of native red-legged frogs (*Rana aurora*). *Conservation Biology* 12:776–787.

Kiesecker, J.M., Blaustein, A.R. and Miller, C.L. 2001a. Potential mechanisms underlying the displacement of native red-legged frogs by introduced bullfrogs. *Ecology* 82: 1964–1970.

King, W.F. 1966. Competition between two south Florida lizards of the genus *Anolis*. *Dissertation Abstracts* 288: 1275.

King, W. 1968. As a consequence many will die. *Florida Naturalist* 41: 99–103, 120.

King, W. and Krakauer. T. 1966. The exotic herpetofauna of southeast Florida. *Quarterly Journal of Florida Academy of Science* 29: 144–154.

Knapp, S. Barry, S. and Brown, L. (2005 in prep.) CLIMATE software manual: Version 2. Bureau of Rural Sciences, Canberra.

Kolar, C.S. and Lodge, D.M. 2001. Progress in invasion biology: predicting invaders. *Trends in Ecology and Evolution* 16: 199–204.

Kolar, C.S. and Lodge, D.M. 2002. Ecological predictions and risk assessment for alien fishes in North America. *Science* 298: 1233–1236.

Kraus, F. 2000. Problematic frogs trouble people Hawaiian Ecosystems At Risk (Pest Alert) Project. http://www.hear.org/AlienSpeciesInHawaii/species/frogs/index.html

Kraus, F. 2003. Invasion pathways for terrestrial vertebrates. Pp. 68–92 in Carlton, J., Ruiz, G. and Mack, R. (eds), *Invasive Species: Vectors and Management Strategies.* Island Press, Washington, DC.

Kraus, F. and Campbell, E.W. 2002. Human-mediated escalation of a formerly eradicable problem: the invasion of Caribbean frogs in the Hawaiian Islands. *Biological Invasions* 4: 327–332.

Kraus, F., Campbell, E.W., Allison, A. and Pratt, T. 1999. *Eleutherodactylus* frog introductions to Hawaii. *Herpetological Review* 30: 21–25.

Kraus, F. and Cravalho, D. 2001. The risk to Hawai'i from snakes. *Pacific Science* 55: 409–417.

Kupferberg, S.J. 1997. Bullfrog (*Rana catesbeiana*) invasion of a California river: The role of larval competition. *Ecology* 78: 1736−1751.

Lafferty, K. D. and Page, C. J. 1997. Predation on the endangered tidewater goby, *Eucyclogobius newberryi*, by the introduced African clawed frog, *Xenopus laevis*, with notes on the frog's parasites. *Copeia* 1997: 589–592.

Lawler, S. P., D. Dritz, T. Strange, and M. Holyoak. 1999. Effects of introduced mosquitofish and bullfrogs on the threatened California red-legged frog. *Conservation Biology* 13:613–622.

Lever, C. 2003. *Naturalized Reptiles and Amphibians of the World*. Oxford University Press, UK.

Lewontin, R.C. and Birch, L.C. 1966. Hybridization as a source of variation for adaptation to new environments. *Evolution* 20: 315–336.

Licht, L.E. 1967. Growth inhibition in crowded tadpoles: intraspecific and interspecific effects. *Ecology* 48: 736–745.

Lidicker, W.Z. 1991. Introduced mammals in California. Pp. 263–271 in: Groves, R.H. and di Castri, F. (eds) *Biogeography of Mediterranean Invasions*. Cambridge University Press, Cambridge.

Lindeman, P.V., 1999. Aggressive interactions during basking among four species of emydid turtles. *Journal of Herpetology* 33: 214–219.

Lobos, G. and Jaksic, F.M. 2005. The ongoing invasion of African clawed frogs (*Xenopus laevis*) in Chile: causes of concern. *Biodiversity and Conservation* 14: 429-439

Lobos, G. and Measey G.J. 2002. Invasive population of *Xenopus laevis* (Daudin) in Chile. *Herpetological Journal* 12: 163–168.

Lodge, D.M. 1993a. Species invasions and deletions: community effects and responses to climate and habitat change. Pp. 367–387 in: P.M. Kareiva, J.G. Kingsolver and R.B. Huey (eds) *Biotic Interactions and Global Change*. Sinauer Associates Inc., Sunderland, Massachusetts.

Lodge, D.M. 1993b. Biological invasions: lessons for ecology. *Trends in Ecology and Evolution* 8: 113–137.

Lodge, D.M. 1993c. Reply from David Lodge. *Trends in Ecology and Evolution* 8: 380–381.

Loope, L.L., Howarth, F.G., Kraus, F. and Pratt, T.K. 2001. Newly emergent and future threats of alien species to Pacific landbirds and ecosystems. *Studies in Avian Biology* 22: 291–304.

Losos, J.B., Marks, J.C. and Schoener, T.W. 1993. Habitat use and ecological interactions of an introduced and a native species of *Anolis* lizard on Grand Cayman, with at review of the outcomes of anole introductions. *Oecologia* 95: 525–532.

Losos, J.B. and Spiller, D.A. 1999. Differential colonization success and asymmetrical interactions between two lizard species. *Ecology* 80: 252–258.

Lowe S., Browne M., Boudjelas S., De Poorter M. (2004) 100 of the World's Worst Invasive Alien Species A selection from the Global Invasive Species Database. Published by The Invasive Species Specialist Group (ISSG) a specialist group of the Species Survival Commission (SSC) of the World Conservation Union (IUCN). Updated and reprinted version: November 2004. http://www.issg.org/booklet.pdf

Luiselli, L., Capula, M., Capizzi, D., Filippi, E., Jesus, V.T. and Ainbaldi, C.1997. Problems for conservation of pond turtles (*Emys obicularis*) in central Italy: is the introduced red-eared turtle (*Trachemys scripta*) a serious threat? *Chelonian Conservation and Biology* 2: 417-419.

MacIsaac, H.J., Grigorovich, I.A. and Ricciardi, A. 2001. Reassessment of species invasions concepts: the Great Lakes basin as a model. *Biological Invasions* 3: 405–416.

Martínez-Morales, M.A. and Cuarón, A.D. 1999. Boa constrictor, an introduced predator threatening the endemic fauna on Cozumel Island, Mexico. *Biodiversity and Conservation* 8: 957–963.

McCann 1996. Nonindigenous Amphibians and Reptiles http://aquat1.ifas.ufl.edu/mcamphib.html

McCoid M.J., 1995. Non-native reptiles and amphibians. Pages 433-437 in LaRoe ET, Farris GS, Puckett CE, Doran PD, Mac MJ, eds. *Our Living Resources: a Report to the Nation on the Distribution, Abundance, and Health of U.S. Plants, Animals and Ecosystems.* Washington, DC.: U.S. Department of the Interior, National Biological Service. [http://biology.usgs.gov/s+t/noframe/x187.htm]

McCoid, M.J. and Fritts, T.H. 1993. Speculations on colonizing success of the African clawed frog, *Xenopus laevis* (Pipidae), in California. *South African Journal of Zoology* 28: 59–61.

McCoid, M.J. and Fritts, T.H. 1995. Female reproductive potential and winter growth of African clawed frogs (Pipidae: *Xenopus laevis*) in California. *California Fish and Game* 81: 39–42.

McKeown, S. 1978. Hawaiian reptiles and amphibians. Oriental Publishing Company Honolulu.

Measey, G.J. 1998a. Terrestrial prey capture in *Xenopus laevis. Copeia* 1998: 787–791.

Measey, G.J. 1998b. Diet of feral *Xenopus laevis* (Daudin) in South Wales. *Journal of Zoology, London* 254: 287–298.

Measey, G.J. 1998c. Growth and aging of feral *Xenopus laevis* (Daudin) in South Wales, U.K. *Journal of Zoology, London* 254: 547–545.

Measey, G.J. and Tinsley, R.C. 1998. Feral *Xenopus* in South Wales. *Herpetological Journal* 8: 23–27.

Melville, J., 2002. Competition and character displacement in two species of scincid lizards. *Ecology Letters* 5: 386–393.

Meshaka, W.E.Jr., Clause, R.M., Butterfield, B.P. and Hauge, J.B. 1997. The Cuban green anole, *Anolis porcatus*, a new anole established in Florida. *Herpetological Review* 28: 101–102.

Meshaka, W.E.Jr., Butterfield, B.P. and Hauge, J.B. 2004. *The Exotic Amphibians and Reptiles of Florida.* Krieger Publishing Company, Malabar, Florida.

Milner, R.L. 1986. Status of the western pond turtle (*Clemys marmorata*) in northwestern Washington. Washington Department of Game, Nongame Division, Olympia, Washington, USA

Mooney. H.A. and Cleland, E.E.. 2001. *Proceedings of the National Academy of Science, USA* 98: 5446–5451.

Moore, R.D., Griffiths, R.A. and Román, A. 2004a. Distribution of the Mallorcan midwife toad (*Alytes muletensis*) in relation to landscape topography and introducted predators. *Biological Conservation* 116: 327–332.

Moore, R.D., Griffiths, R.A. O-Brien, C.M., Murphy, A. and JayY, D. 2004b. Induced defences in an endangered amphibian in response to an introduced snake predator. *Oecologia* 141: 139–147.

Morey, S.R. and Guinn, D.A. 1987. Activity patterns, food habits, and changing abundance in a community of vernal pool amphibians. Pp 149–158 in Williams, D.F., Byrne, S. and Rado, T.A. (eds). Endangered and sensitive species of the San Joaquin Valley, California. California Energy Commission, Sacremento, California, USA.

Moyle, P.B. 1973. Effects of introduced bullfrogs, *Rana catesbeiana*, on the native frogs of the San Joaquin Valley, California. *Copeia* 1973: 18–22.

Moyle, P.B. 1973. Effects of introduced bullfrogs, *Rana catesbeiana*, on the native frogs of the San Joaquin Valley, California. *Copeia* 1973: 18–22.

Moyle, P.B. and Light, T. 1996a. Fish invasions in California: do abiotic factors determine success? *Ecology* 77: 1666–1670.

Moyle, P.B. and Light, T. 1996b. Biological invasions of fresh water: empirical rules and assembly theory. *Biological Conservation* 78: 149–161.

Nevo, E., Gorman, G., Soule, M., Yang, S.Y., Clover, R. and Jovanovi.V. 1972. Competitive exclusion between insular *Lacerta* species (Sauria: Lacertilia). *Oecologia* 10: 183–190.

Newberry, R. 1984. The American red-eared terrapin in South Africa. *African Wildlife* 38: 186–189.

Norton, T., Nix, H. and Williams, J. 1996. Risk, uncertainty and cumulative environmental change. Pp. 33–43 in: Norton, T.W., Beer, T. and Dovers, S.R. (eds) *Risk and Uncertainty in Environmental Management — 1995 Fenner Conference on the Environment.* Australian Academy of Science, Canberra.

Pacala, S. and Roughgarden, J. 1982. Resource partitioning and interspecific competition in two two-species insular *Anolis* lizard communities. *Science* 217: 444–446.

Parker, I.M., Simberloff, D., Lonsdale, W.M., Goodell, K., Wonham, M. Kareiva, P.M., Williamson, M.H., von Holle, B., Byers, J.E. and Goldwasser, L. 1999. Impact: towards a framework for understanding the ecological effects of invaders. *Biological Invasions* 1: 3–19.

Perry, G. and Morton, J.M. 1999. Regeneration rates of the woody vegetation of Guam's Northwest Field following major disturbance: land use patterns, feral ungulates, and cascading effects of the brown treesnake. *Micronesica* 31: 125–142.

Petranka, J.W. 1989. Chemical interference competition in tadpoles: does it occur outside laboratory aquaria? *Copeia* 1989: 921–930.

Petren, K. and Case, T.J. 1996. An experimental demonstration of exploitation competition in an ongoing invasion. *Ecology* 77: 118–132.

Petren, K. and Case, T.J., 1998. Habitat structure determines competition intensity and invasion success in gecko lizards. *Proceedings of the National Academy of Sciences of the United States of America* 95: 11739–11744.

Phillips, B.L., Brown, G.P. and Shine, R. 2003. Assessing the potential impact of cane toads on Australian snakes. *Conservation Biology* 17: 1738–1747.

Phillips, B.L. and Shine, R. 2004. Adapting to an invasive species: Toxic cane toads induce morphological change in Australian snakes. *Proceedings of the National Academy of Sciences (USA)* 101: 17150–17155.

Powell, R., Smith, D.D. Parmerlee, J.S. Jr., Taylor, C.V. and Jolley, M.L. 1990. Range expansion by an introduced anole: *Anolis porcatus* in the Dominican Republic. *Amphibia–Reptilia* 11: 421–425.

Rabor, D.S. 1952. Preliminary notes on the giant toad, *Bufo marinus* (Linn.), in the Philippine Islands. *Copeia* 1952: 281–282.

Radovanovic, M., 1965. Experimentelle beiträge zum problem der kompetition. *Zoologicher Anzeiger* 29: 534–539.

Reagan, D.P. 1992. Congeneric species distribution and abundance in a three-dimensional habitat: the rain forest anoles of Puerto Rico. *Copeia* 1992: 392–403.

Rhymer, J.M. and Simberloff, D. 1996. Extinction by hybridization and introgression. *Annual Review of Ecology and Systematics* 27: 83–109.

Ricciardi, A. 2003. Predicting the impacts of an introduced species from its invasion history: an empirical approach applied to zebra mussel invasions. *Freshwater Biology* 48: 972–981.

Ricciardi, A. and Rasmussen , J.B. 1998. Predicting the identity and impact of future biological invaders: a priority for aquatic resource management. *Canadian Journal of Fisheries and Aquatic Sciences* 55: 1759–1765.

Riley, S.P.D., Shaffer, H.B., Voss, S.R. and Fitzpatrick, B.M. 2003. Hybridization between a rare native salamander (*Ambystoma californiense*) and its introduced congener. *Ecological Applications* 13: 1263–1275.

Rodda, G.H. and Fritts, T.H. 1992. The impact of the introduction of the colubrid snake *Boiga irregularis* on Guam's lizards. *Journal of Herpetology* 26: 166–174.

Rodda, G.H., Fritts, T.H. and Chiszar, D. 1997. The disappearance of Guam's wildlife: new insights for herpetology, evolutionary ecology, and conservation. *BioScience* 47: 565–574.

Rodda, G.H. McCoid, M.J., Fritts, T.H. Campbell III, E.W. 1999. Population trends and limiting factors in Boiga irregularis. Pp 236–253 in Rodda, G.H., Sawai, Y., Chiszar, D. and Tanaka, H. (Eds) Problem snake management: the habu and brown tree snake. Cornell University Press, Ithaca, NY.

Rodda, G.H., Perry, G., Rondeau, R.J. and Laxell, J. 2001. The densest terrestrial vertebrate. *Journal of Tropical Ecology* 17: 33–338.

Rosen, P.C. and Schwalbe, C.R. 1995. Bullfrogs (*Rana catesbeiana*): introduced predators in southwestern wetlands. Pages 452–454 in Laroe, E. T., Farris, G. S., Puckett, C. E., Doran, P. D. and Mac, M. J. (eds). Our living resources: a report to the nation on the distribution, abundance, and health of U.S. plants, animals, and ecosystems. U.S. Department of the Interior, National Biological Service, Washington, D.C. http://biology.usgs.gov/s+t/noframe/x188.htm

Rosen, P.C., Schwalbe, C.R. and Sartorius, S.S. 1996. Decline of the Chiricahua leopard frog in Arizona mediated by introduced species. Report to Heritage program, Arizona Game and Fish Department.

Rossi, J.V. 1981. *Bufo marinus* in Florida: some natural history and its impact on native vertebrates. Master's thesis. Department of Botany. University of South Florida.

Savidge, J.A. 1987. Extinction of an island avifauna by an introduced snake. *Ecology* 68: 660–668.

Schoener, T.W. and Spiller, D.A. 1996. Devastation of prey diversity by experimentally introduced predators in the field. *Nature* 381: 691–694.

Schoener, T.W. and Spiller, D.A. 1999. Indirect effects of an experimentally staged invasion by a major predator. *American Naturalist* 153: 347–358.

Schwalbe, C.R. and Rosen, P.C. 1988. Preliminary report on effects of bullfrogs on wetland herpetofaunas in Southeastern Arizona.. Pages 166–173 in Szaro, R.C., Steverson, K.E. and Patton, J.R. (eds). Management of amphibians, reptiles and small mammals in North America. U.S. Department of Agriculture, Forest Service, Flagstaff, AZ, USA.

Searle, D.B. 1980. Influence of amphibian larvae on primary production, nutrient flux, and competition in a pond ecosystem. *Ecology* 61: 1531–1550.

Shine S., Williams, N. and Gündling, L. 2000. A Guide to Designing Legal Institutional Frameworks on Alien Invasive Species. IUCN, Gland, Switzerland, Cambridge and Bonn. (English version only). Hardcopies of this publication are available in English, French and Spanish and can be obtained from: IUCN Publications Services Unit. http://www.issg.org/

Simberloff, D. 1989. Which insect introductions succeed and which fail? Pp. 61–75 in: Drake, J.A., Mooney, H.A., di Castri, F., Groves, R.H., Kruger, F.J., Rejmanek, M. and Williamson, M.W. (eds) *Biological Invasions. A Global Perspective*. John Wiley and Sons, Chichester.

Simberloff, D. 1995. Why do introduced species appear to devastate islands more than mainland areas? *Pacific Science* 49: 87–97.

Simberloff, D. 1997–1998. Facing the future. *World Conservation* 4/97–1/98: 21–23.

Smith, C.S., Lonsdale, W.M. and Fortune, J. 1999. When to ignore advice: invasion predictions and decision theory. *Biological Invasions* 1: 89–96.

Smith, K.G. 2005. Effects of nonindigenous tadpoles on native tadpoles in Florida: evidence of competition. *Biological Conservation* 123: 433–441.

Smith, K.G. in press 2005. An exploratory assessment of Cuban treefrog (*Osteopilus septentrionalis*) tadpoles as predators of native and nonindigenous tadpoles in Florida. *Amphibia–Reptilia*.

Speare, R. 1990. A review of the diseases of the cane toad, *Bufo marinus*, with comments on biological control. *Australian Wildlife Research* 17: 387–410.

Spiller, D.A. and Schoener, T.W. 1997.Folivory on islands with and without insectivorous lizards: an eight-year study. *Oikos* 78: 15–22.

Spiller, D.A. and Schoener, T.W. 1998. Lizards reduce spider species richness by excluding rare species. *Ecology* 79: 503–516.

Spinks, P.Q., Pauly, G.B., Crayon, J.J. and Shaffer, H.B. 2003. Survival of the western pond turtle (*Emys marmorata*) in an urban California environment. *Biological Conservation* 113: 257–267.

Stewart, M.M. and Rand, A.S. 1991. Vocalizations and the defence of retreat sites by male and female frogs, *Eleutherodactylus coqui*. *Copeia* 1991:1013–1024.

Storfer, A., Mech, S.G., Reudink, M.W., Ziemba, R.E., Warren, J. and Collins, J.P. 2004. Evidence for introgression in the endangered Sonora Tiger Salamander, *Ambystoma tigrinum stebbinsi* (Lowe). *Copeia* 2004: 783–796.

Stumpel, A.H.P. 1992. Successful reproduction of introduced bullfrogs *Rana catesbeiana* in northwestern Europe: a potential threat to indigenous amphibians. *Biological Conservation* 66: 61–62.

Taylor, J.N., Courtenay Jr, W.R. and Mc Cann, J.A. 1984. Known impacts of exotic fishes in the continental United States. Pp. 322–373 in: Courtenay Jr, W.R. and Stauffer Jr, J.R. (eds) *Distribution, Biology and Management of Exotic Fishes.* John Hopkins University Press, Baltimore.

Tinsley, R.C. and McCoid, M.J. 1996. Feral populations of *Xenopus* outside Africa. *Symposia of the Zoological Society of London* 68: 81–94.

Tonge, S. 1986. Collecting the Mallorcan midwife toad. *Oryx* 20: 74–78.

Townsend, C. R. 1991. Exotic species management and the need for a theory of invasion ecology. *New Zealand Journal of Ecology* 15: 1–3.

Townsend, C.R. and Winterbourn, M.J. 1992. Assessment of the environmental risk posed by an exotic fish: the proposed introduction of channel catfish (*Ictalurus punctatus*) to New Zealand. *Conservation Biology* 6: 273–282.

Tummons, P. 2003. More Than a Noisy Nuisance, Coqui Are Eyed as Threat to Island Fauna. *Environment Hawai`i, Inc.* 13 (12). http://www.environment-hawaii.org/603more.htm

Tyler, M. 2001. Introduced Amphibians and their impact on native wildlife. Proceedings of Exotic Reptile & Amphibian Conference, October 2001, Barossa Valley, South Australia. http://homepages.ihug.com.au/~venomsup/exoticconference/miketylerstalk1.html

United States Geological Survey (USGS). 2002. Nonindigenous Amphibian Distribution Information. http://nas.er.usgs.gov/amphibians/index.html.

United States Geological Survey (USGS). 2003a. Impact of invasive exotic animals on native reptiles and amphibians -- Slider turtle. http://www.werc.usgs.gov/hq/invasive/slider.html

United States Geological Survey (USGS). 2003b. Impact of invasive exotic animals on native reptiles and amphibians -- Brown tree snake. http://www.werc.usgs.gov/hq/invasive/snake.html

United States Geological Survey (USGS). 2003c Impact of invasive exotic animals on native reptiles and amphibians -- African clawed frog. http://www.werc.usgs.gov/hq/invasive/clawfrog.html

United States Geological Survey (USGS). 2003d. Impact of invasive exotic animals on native reptiles and amphibians -- Cane toad / Bullfrog. http://www.werc.usgs.gov/hq/invasive/toad-frog.html

Vincent, T.C. 2002. A field and enclosure study of population dynamics and the effects of density on fecundity of *Anolis carolinensis* and *Anolis sagrei* (Sauria: Polychrotidae) in conspecifics and heterospecific groups. PhD thesis, Department of Ecology and Evolutionary Biology, University of Tennessee, Knoxville, Tennessee.

Vitousek, P.M., D'Antonio, C.M., Loope, L.L., Rejmanek, M. and Westbrooks, R. (1997) Introduced species: a significant component of human-caused global change. *New Zealand Journal of Ecology* 21: 1–16.

Vitousek, P.M., Loope, L.L., and Stone, C.P. 1987. Introduced species in Hawaii: biological effects and opportunities for ecological research. *Trends in Ecology and Evolution* 2: 224–227.

Waide, R.B. and Reagan, D.P. 1983. Competition between West Indian anoles and birds. *American Naturalist* 121: 133–138.

Welcomme, R.L. 1988. *International Introductions of Inland Aquatic Species*. Food and Agriculture Organisation Fisheries Technical Paper 294. FAO United Nations, Rome.

Weldon, C., du Preez, L.H., Hyatt, A.D., Muller, R. and Speare, R. 2004. Origin of the Amphibian Chytrid Fungus. *Emerging Infectious Diseases* 10: 2100–2105.

Werner, E.E. 1994. Ontogenetic scaling of competitive relations–size-dependent effects and responses in two anuran larvae. *Ecology* 75: 197–213.

West, J.A. 1979. The occurrence and spread of some exotic reptiles and amphibians in New Zealand. *Herpetofauna* 10: 4–9.

Wiles, G.J., Bart, J., Beck, R.E. and Aguon, C.F. 2003. Impacts of the Brown Tree Snake: Patterns of Decline and Species Persistence in Guam's Avifauna. *Conservation Biology* 17: 1350–1360.

Williamson, M. 1989. Mathematical models of invasion. Pp. 329–350 in: Drake, J.A., Mooney, H.A., di Castri, F., Groves, R.H., Kruger, F.J., Rejmanek, M. and Williamson, M.W. (eds) *Biological Invasions. A Global Perspective*. John Wiley and Sons, Chichester.

Williamson, M. 1996. *Biological Invasions*. Chapman and Hall, London.

Williamson, I. 1999. Competition between the larvae of the introduced cane toad *Bufo marinus* (Anura: Bufonidae) and native anurans from the Darling Downs area of southern Queensland. *Australian Journal of Ecology* 24: 636–643.

Williamson, M.H. and Brown, K.C. 1986. The analysis and modelling of British invasions. *Philosophical Transactions of the Royal Society, London B* 314: 505–522.

Williamson, M. and Fitter, A. 1996. The varying success of invaders. *Advances in Invasion Ecology* 77: 1651–1666.

Wilson, L. D. and L. Porras. 1983. The ecological impact of man on the south Florida herpetofauna. University of Kansas Museum of Natural History Special Publication No. 9:1–89.

Zug, G.R., Lindgren, E. and Pippet, J.R. 1975. Distribution and ecology of the Marine Toad, *Bufo marinus*, in Papua New Guinea. *Pacific Science* 29: 31–50.

Appendix A

Factors affecting establishment success of exotic reptiles and amphibians

Table A1. Exotic reptiles and amphibians introduced to Britain. Taxonomic scores, climate match scores, successful introduction elsewhere scores, geographic range sizes and Establishment Risk Scores[1,2]

Introduction outcome	**Family**	**Taxonomic score**	**Climate matches** **Σ7–10**[3]	**Climate Match Score** (% of 194)[4]	**Geographic range size outside Britain** (million km^2)	**Successful Elsewhere Score**	**Establishment Risk Score**
Britain successful species							
Alytes obstetricians (mid-wife toad)	Discoglossidae	15	190	98	2.5	30	143
Rana lessonae (pool frog)	Ranidae	30	192	99	9.2	30	159
Rana ridibunda (marsh frog)	Ranidae	30	189	97	7.8	30	157
Trituris alpestris (alpine newt)	Salamandridae	15	190	98	3.7	30	143
Triturus carnifex (Italian crested newt)	Salamandridae	15	101	52	0.2	30	97
Xenopus laevis (African clawed toad)	Pipidae	15	94	48	12.5	30	93
Elaphe longissima (Aesculapian snake)	Elapidae	10	161	83	2.4	0	93
Podarcis muralis (common wall lizard)	Lacertidae	20	190	98	4.0	30	148
Average Britain successful species		**18.8**	**163**	**84.2**	**5.26**	**26.3**	**129**
Britain failed species							
Salamandra salamandra (European fire salamander)	Salamandridae	15	190	98	7.6	0	113
Hydromantes genei (Sardinian cave salamander)	Plethodontidae	20	0	0	0.0004	0	20
Chalcides ocellatus (ocellated skink)	Scincidae	15	5	3	16.5	30	48
Eleutherodactylus johnstonei (piping frog)	Leptodactylidae	30	0	0	0.0009	30	60
Discoglossus pictus (painted frog)	Discoglossidae	15	76	39	0.3	30	84
Bombina bombina (fire-bellied toad)	Discoglossidae	15	121	62	7.0	0	77

Table A1. Cont.

Bufo viridus (green toad)	Bufonidae	20	134	69	12.5	30	119
Hyla aborea (European tree frog)	Hylidae	15	192	99	6.1	0	114
Hyla meridionalis (stripeless tree frog)	Hylidae	15	64	33	0.8	30	114
Pseudacris (Hyla) regilla (Pacific tree frog)	Hylidae	15	45	23	2.5	30	68
Litoria ewingii (brown tree frog)	Hylidae	15	185	95	0.6	30	140
Pelobates fuscus (common spadefoot toad)	Pelobatidae	0	188	97	13.2	0	97
Rana pipiens (northern leopard frog)	Ranidae	30	0	0	14.3	30	60
Scinax rubra (red-snouted tree frog)	Hylidae	15	0	0	4.3	30	45
Chelydra serpentia (common snapping turtle)	Chelydridae	10	0	0	9.4	30	40
Chrysemys picta (painted turtle)	Emydidae	15	0	0	10.9	15	30
Emys orbicularis (European pond turtle)	Emydidae	15	172	89	9.2	30	134
Pelodiscus sinensis (common soft-shell turtle)	Trionychidae	20	0	0	2.8	30	50
Terrapene carolina (eastern box turtle)	Emydidae	15	0	0	1.5	30	45
Testudo graeca (spur-thighed turtle)	Testudinidae	15	3	2	3.1	30	47
Pseudocordylus microlepidotus (Cape Crag lizard)	Cordylidae	10	0	0	0.2	0	10
Tarentola delalandii (Tenerife wall gecko)	Gekkonidae	30	0	0	0.03	0	30
Tarentola mauritanica (Moorish gecko)	Gekkonidae	30	0	0	1.1	30	60
Podarcis dugesii (Madeira wall lizard)	Lacertidae	20	0	0	0.07	30	50
Podarcis sicula (Italian wall lizard)	Lacertidae	20	0	0	0.3	30	50
Lacerta bilineata (green lizard)	Lacertidae	20	180	93	3.5	30	143
Lacerta lepida (ocellated lizard)	Lacertidae	20	108	56	1.4	0	76
Natrix maura (viperine snake)	Colubridae	10	129	66	1.5	30	106
Natrix tessellata (tessallated snake)	Colubridae	10	91	47	13.0	0	57

Table A1. Cont.

Thamnophis sirtalis (common garter snake)	Colubridae	10	0	0	16.0	0	10
Lampropeltis triangulum (milk snake)	Colubridae	10	8	04	9.7	0	14
Coluber jugularis (large whip snake)	Colubridae	10	3	02	2.3	0	12
Average Britain failed species		**16.4**	**59**	**30.5**	**5.4**	**18.3**	**66**

[1] The following species were excluded from the table because their status in Britain is uncertain: *Bombina variegata*; *Rana catesbeiana*; *Trachemys scripta.*

[2] *Rana esculenta* was excluded from the table because it is a hybrid between *R. lessonae* and *R. ridibunda* (Lever 2003).[R. esculenta should not be excluded. It is not a hybrid as the term is usually understood but is a curious type of taxon referred to as a klepton, which behaves much like a species and has a huge range across Europe.

[3] T-test *P* value(successful spp vs failed spp) CLIMATE match Σ7–10 = 0.000279 (very highly significant).

[4] 194 is the highest possible CLIMATE match Σ7–10 score possible for Britain.

Table A2. Exotic and translocated reptiles and amphibians introduced to California[1]. Taxonomic scores, climate match scores, history of successful introductions, geographic range sizes and Establishment Risk Scores

Introduction outcome	**Family**	**Taxonomic score**	**Climate matches** Σ7–10[2]	**Climate Match Score** (% of 172)[3]	**Geographic range size outside California** (million km²)	**Successful Elsewhere Score**	**Establishment Risk Score**
California successful species							
Xenopus laevis (African clawed toad)	Pipidae	15	72	42	12.5	30	87
Chamaeleo jacksonii (Jackson's chameleon)	Chamaeleonidae	30	0	0	0.8	30	60
Hemidactylus turcicus (Mediterranean gecko)	Gekkonidae	30	138	80	17.1	30	140
Tarentola mauritanica (Moorish gecko)	Gekkonidae	30	87	51	1.1	30	111
Ambystoma tigrinum (tiger salamander)	Ambystomatidae	15	24	14	15.1	30	59
Rana berlandieri (Rio Grande leopard frog)	Ranidae	30	10	6	0.9	30	66
Rana catesbeiana (American bullfrog)	Ranidae	30	116	67	10.7	30	127
Apalone spinifera (spiny softshell turtle)	Trionychidae	20	19	11	6.8	30	61
Chelydra serpentina (common snapping turtle)	Chelydridae	10	1	1	9.4	30	41
Trachemys scripta (red-eared slider)	Emydidae	15	22	13	4.5	30	58
Nerodia fasciata (southern water snake)	Colubridae	10	0	0	2.1	30	40
Average California successful species		**21.4**	**44.45**	**26**	**7.36**	**30**	**77.3**
California failed species							
Andrias japonicus (Japanese giant salamander)	Cryptobranchidae	0	0	0	0.006	0	0
Hemidactylus garnotii (Indo-Pacific gecko)	Gekkonidae	30	0	0	2.9	30	60
Hemiphyllodactylus typus (Indo-Pacific tree gecko)	Gekkonidae	30	0	0	0.01	30	60
Gehyra mutilata (stump-toed gecko)	Gekkonidae	30	0	0	0.03	30	60

Table A2. Cont.

Heloderma horridum (Mexican beaded lizard)	Helodermatidae	0	0	0	0.9	0	0
Cordylus giganteus (giant sungazer lizard)	Cordylidae	10	0	0	0.006	0	10
Ctenosaura hemilopha (black iguana)	Iguanidae	20	0	0	0.03	30	50
Lepidodactylus lugubris (mourning gecko)	Gekkonidae	30	60	35	1.4	30	95
Iguana iguana (common iguana)	Iguanidae	20	0	0	14	30	50
Palea steindachneri (wattle-necked softshell turtle)	Trionychidae	20	0	0	4.5	30	50
Geochelone carbonaria (red footed tortoise)	Testudinidae	15	0	0	0.2	0	15
Varanus salvator (water monitor)	Varanidae	15	0	0	5.2	0	15
Chinemys reevesii (Reeve's turtle)	Geoemydidae	0	8	5	4	0	5
Lamprophis fuliginosus (African house snake)	Colubridae	10	51	3	12.5	0	40
Leptodeira annulata (banded cat-eyed snake)	Colubridae	10	0	0	14.1	0	10
Naja haje (Egyptian cobra)	Elapidae	10	103	6	23.1	0	70
Boa constrictor (boa constrictor)	Boidae	5	4	2	17.3	30	37
Corallus hortulanus (Amazon tree boa)	Boidae	5	0	0	9.2	0	5
Python reticulatus (reticulated python)	Boidae	5	0	0	2.4	0	5
Python molurus (Burmese python)	Boidae	5	0	0	4.6	30	35
Caiman crocodilus (spectacled caiman)	Alligatoridae	15	0	0	10.4	30	45
Notophthalmus viridescens (eastern newt)	Salamandridae	15	0	0	10.5	0	15
Bufo marinus (cane toad)	Bufonidae	20	1	1	16.2	30	51
Hyla wrightorum (Madrean tree frog)	Hylidae	15	5	3	0.2	0	18
Anolis carolinensis green anole)	Iguanidae	20	1	1	0.8	30	51
Eumeces obsoletus (great plains skink)	Scincidae	15	1	1	2.3	0	16
Sceloporus poinsettii (crevice spiny lizard)	Lacertidae	20	1	1	0.9	0	21
Sceloporus jarrovii (moutain Spiny lizard)	Lacertidae	20	1	1	0.3	0	21

Table A2. Cont.

Sceloporus serrifer (blue spiny lizard)	Lacertidae	20	0	0	0.9	0	20
Phrynosoma cornutum (Texas horned lizard)	Lacertidae	20	1	1	2.8	30	51
Pseudemys floridana (common cooter)	Emydidae	15	0	0	0.4	0	15
Pseudemys concinna (river cooter)	Emydidae	15	0	0	1.9	30	45
Graptemys pseudogeographica (False Map Turtle)	Emydidae	15	0	0	4.2	30	45
Macrochelys temminckii (alligator snapping turtle)	Chelydridae	10	0	0	2.4	0	10
Malaclemys terrapin (diamondback turtle)	Emydidae	15	0	0	0.4	15	30
Terrapene carolina (eastern box turtle)	Emydidae	15	0	0	1.5	30	45
Terrapene ornata (ornate box turtle)	Emydidae	15	10	6	2.2	0	21
Elaphe guttata (corn snake)	Elapidae	10	1	1	5.7	30	41
Drymarchon corais (indigo snake)	Colubridae	10	0	0	1	0	10
Lampropeltis triangulum (milk snake)	Colubridae	10	4	2	9.7	0	12
Nerodia sipedon (northern water snake)	Colubridae	10	0	0	4.9	0	10
Opheodrys aestivus (rough green snake)	Colubridae	10	0	0	8.4	0	10
Thamnophis sauritus (eastern ribbon snake)	Colubridae	10	0	0	3.1	15	25
Alligator mississipiensis (American alligator)	Alligatoridae	10	0	0	1.3	15	25
Average California failed species		**14.1**	**5.73**	**3**	**4.75**	**12.6**	**30.1**

[1] Translocated reptiles and amphibians introduced to California from elsewhere in the United States were included in this dataset to increase sample sizes.

[2] T-test *P* value(successful spp vs failed spp) CLIMATE match Σ7–10 = 0.003869 (highly significant).

[3] 172 is the highest possible CLIMATE match Σ7–10 score possible for California.

Table A3. Exotic reptiles and amphibians introduced to Florida. Taxonomic scores, climate match scores, history of successful introductions, geographic range sizes and Establishment Risk Scores[1,2]

Introduction outcome	**Family**	**Taxonomic score**	**Climate matches** $\Sigma 7\text{–}10$[3]	**Climate Match Score** (% of 106)[4]	**Geographic range size outside Florida** (million km^2)	**Successful Elsewhere Score**	**Establishment Risk Score**
Florida successful species							
Eleutherodactylus coqui (Puerto Rican coqui)	Leptodactylidae	30	1	1	0.0003	30	61
Eleutherodactylus planirostris (greenhouse frog)	Leptodactylidae	30	76	72	0.06	30	132
Osteopilus septentrionalis (Cuban tree frog)	Hylidae	15	45	42	0.0003	30	86
Ramphotyphlops braminus (flowerpot snake)	Typhloidae	30	89	84	7.5	30	144
Boa constrictor (boa constrictor)	Boidae	5	100	94	17.3	30	129
Python molurus (Burmese python)	Boidae	5	68	64	4.6	0	69
Chamaeleo calyptratus (veiled chameleon)	Chamaeleonidae	30	0	0	0.1	30	60
Agama agama (African red-headed agama)	Agamidae	30	11	1	9.2	30	61
Calotes versicolor (bloodsucker lizard)	Agamidae	30	70	66	3.9	30	126
Ctenosaura pectinata (Mexican spiny-tailed iguana)	Iguanidae	20	0	0	0.003	30	50
Ctenosaura similis (black spiny-tailed iguana)	Iguanidae	20	7	7	0.1	0	27
Iguana iguana (common green iguana)	Iguanidae	20	80	75	14.0	30	125
Leiocephalus carinatus (Northern curly-tailed lizard)	Iguanidae	20	57	54	0.0001	0	74
Leiocephalus schreibersi (red-sided curly-tailed lizard)	Iguanidae	20	5	5	0.0003	0	25
Anolis chlorocyanus (Haitian green anole)	Iguanidae	20	8	8	0.001	30	58
Anolis cybotes (large-headed anole)	Iguanidae	20	8	8	0.001	30	58
Anolis distichus (Bahamian bark anole)	Iguanidae	20	29	27	0.0003	30	77
Anolis equestris (knight anole)	Iguanidae	20	38	36	0.0003	30	86
Anolis garmani (Jamaican giant anole)	Iguanidae	20	51	48	0.0006	15	83

Table A3. Cont.

Anolis porcatus (Cuban green anole)	Iguanidae	20	45	42	0.003	30	92
Anolis sagrei (brown anole)	Iguanidae	20	104	98	0.009	30	148
Anolis ferreus (Marie Galante sail-tailed anole)	Iguanidae	20	3	3	0.00006	0	23
Anolis cristatellus (crested anole)	Iguanidae	20	7	7	0.0006	30	57
Basiliscus vittatus (northern brown basilisk)	Iguanidae	20	20	19	0.3	0	39
Cosymbotus platyurus (Asian flat-tailed gecko)	Gekkonidae	30	78	74	2.1	0	104
Gekko gecko (Tokay gecko)	Gekkonidae	30	77	73	4.1	15	118
Gonatodes albogularis (yellow-headed gecko)	Gekkonidae	30	46	43	0.4	30	103
Hemidactylus frenatus (common house gecko)	Gekkonidae	30	76	72	5.7	30	132
Hemidactylus garnotii (Indo-Pacific gecko)	Gekkonidae	30	89	84	2.9	30	144
Hemidactylus mabouia (tropical house gecko)	Gekkonidae	30	37	35	13.4	30	95
Hemidactylus turcicus (Mediterranean house gecko)	Gekkonidae	30	78	74	17.0	30	134
Pachydactylus bibronii (Bibron's gecko)	Gekkonidae	30	4	4	0.9	0	34
Sphaerodactylus elegans (Ashy gecko)	Gekkonidae	30	45	42	0.002	0	72
Tarentola annularis (white-spotted wall gecko)	Gekkonidae	30	4	4	15.3	0	34
Tarentola mauritanica (Moorish gecko)	Gekkonidae	30	0	0	1.21	30	60
Phelsuma madagascariensis (giant day gecko)	Gekkonidae	30	7	7	0.6	30	67
Ameiva ameiva (South American ground lizard)	Teiidae	20	54	51	3.6	0	71
Aspidoscelis motaguae (whiptail lizard)	Teiidae	20	0	0	0.09	0	20
Cnemidophorus lemniscatus (rainbow whiptail)	Teiidae	20	4	4	2.0	30	54
Mabuya multifasciata (east Indian brown skink)	Scincidae	15	89	84	5.1	30	129
Caiman crocodilus (spectacled caiman)	Alligatoridae	10	8	8	10.4	30	48
Average for Florida successful species		**23.2**	**39.46**	**37**	**3.46**	**20.5**	**80.71**

Table A3. Cont.

Florida failed species							
Cynops pyrrhogaster (Japanese fire salamander)	Salamandridae	15	0	0	0.0001	0	15
Atelopus zeteki (Panamanian golden frog)	Bufonidae	20	0	0	0.0001	0	20
Bufo arenarum (common toad)	Bufonidae	20	80	75	1.9	0	95
Bufo blombergi (Columbian giant toad)	Bufonidae	20	0	0	0.09	0	20
Bufo paracnemis (Cururu toad) [Analysed *B. schneideri*]	Bufonidae	20	97	92	2.6	0	112
Pachymedusa dacnicolor (Mexican leaf frog)	Hylidae	15	0	0	0.01	0	15
Hymenochirus boettgeri (Zaire dwarf clawed frog	Pipidae	15	3	3	1.4	0	18
Xenopus laevis (African clawed toad)	Pipidae	15	26	25	12.5	30	70
Typhlops lumbricalis (Diad's blind snake)	Typhlopidae	30	45	42	0.003	0	72
Python regius (royal python)	Boidae	5	0	0	3.0	0	5
Python reticulatus (reticulated python)	Boidae	5	9	8	2.4	0	13
Cordylus cordylus (Cape girdled lizard)	Cordylidae	10	0	0	2.2	0	10
Hemidactylus brookii (Indian house gecko)	Gekkonidae	30	74	7	9.1	30	67
Sphaerodactylus macrolepis (common dwarf gecko)	Gekkonidae	30	3	3	0.3	0	33
Basiliscus basiliscus (common basilisk)	Iguanidae	20	0	0	0.4	0	20
Cyclura cornuta (rhinoceros iguana)	Iguanidae	20	8	8	0.003	0	28
Tupinambis nigropunctatus (black tegu)	Teiidae	20	58	55	8.4	0	75
Chelus fimbriatus (matamata)	Chelidae	10	16	15	3.7	0	25
Trachemys dorbigni (Brazilian slider)	Emydidae	15	1	1	0.8	0	16
Trachemys stejnegeri (Puerto Rican slider)	Emydidae	15	7	7	0.009	30	52
Kinosternon scorpioides (scorpion mud turtle)	Kinosternidae	0	25	24	3.0	0	24
Podocnemis lewyana (Magdalena river turtle)	Pelomedusidae	10	0	0	0.5	0	10
Podocnemis sextuberculata (Amazon river turtle)	Pelomedusidae	10	0	0	2.7	0	10
Podocnemis unifilis (yellow-spotted river turtle)	Pelomedusidae	10	29	27	5.2	0	37
Anolis conspersus (blue-throated anole)	Iguanidae	20	6	6	0.00003	0	26

Table A3. Cont.

Varanus salvator (water monitor)	Varanidae	15	44	42	4.5	0	57
Varanus exanthematicus (savanah monitor)	Varanidae	15	5	5	6.7	0	20
Average for Florida failed species		**15.9**	**19.85**	**19**	**2.65**	**3.3**	**35.74**

[1] Sources: Fred Kraus database of published records; Kevin M. Enge (Florida Fish and Wildlife Conservation Commission, pers. comm. 15 March 2005) list of exotic species established in Florida for at least 10 years; Meshaka et al. (2004).

[2] The following species were excluded from the table because their status in Florida is uncertain, or because it is uncertain whether or not the species has been released in Forida: *Pelodryas caerulea*; *Eleutherodactylus portoricensis*; *Pelusios subniger*; *Geochelone carbonaria*; *Trachemys callirostris*; *Platemys platycephalus*; *Laudakia stellio*; *Leiocephalus personatus*; *Anolis extremus*; *Calotes mystaceus*; *Basiliscus plumifrons*; *Gehyra mutilata*; *Lepidactylus lugubris*; *Ptyodactylus hasselquisti*; *Sphaerodactylus argus*; *Varanus niloticus*; *Acrochordus javanicus*; *Python reticulatus*; *Python sebae*; *Typhlops pusillus*; *Leiolepis belliana*

[3] T-test *P* value (successful spp vs failed spp) CLIMATE match Σ7–10 = 0.008339 (highly significant).

[4] 106 is the highest possible CLIMATE match Σ7–10 score possible for Florida.

Table A4. Exotic reptiles and amphibians introduced to Australia. Taxonomic scores, climate match scores, history of successful introductions, geographic range sizes and Establishment Risk Scores

Introduction outcomes Australian mainland	**Family**	**Taxonomic score**	**Climate matches** $\Sigma 7–10$	**Climate Match Score** % 2785[1]	**Geographic range size outside Australia** (million km^2)	**Successful Elsewhere Score**	**Establishment Risk Score**
Successful species							
Bufo marinus (cane toad)	Bufonidae	20	1849	66	16.2	30	116 (Extreme)
Hemidactylus frenatus (Asian house gecko)	Gekkonidae	30	698	25	5.1	30	85 (Very high)
Lepidodactylus lugubris (mourning gecko)	Gekkonidae	30	79	3	1.3	30	63 (High)
Trachemys scripta (red-eared slider)	Emydidae	15	1504	54	4.5	30	99 (Very high)
Ramphotyphlops braminus (flowerpot snake)	Typhlopidae	30	947	34	7.2	30	94 (Very high)
Average for Australian successful species		**25.0**	**1015**	**36.4**	**6.9**	**30**	**91.4**
Failed species							
Ambystoma mexicanum (axolotl or salamander)	Ambystomatidae	15	0	0	0.0003	0	15 (Very low)
Bufo melanostictus (black-spined toad)	Bufonidae	20	967	35	6.6	30	85 (Very high)
Average for Australian failed species		**17.5**	**483.5**	**17.5**	**3.3**	**15**	**50**
Successful species on Christmas Island							
Lycodon capucinus (wolf snake)	Colubridae	10	616	22	3.6	30	62 (High)
Lygosoma bowringii (grass-skink)	Scincidae	15	775	28	5.4	0	43 (Low)

[1] 2785 is the highest possible CLIMATE match Σ7–10 score possible for Australia.