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FINAL REPORT FOR ENVIRONMENT AUSTRALIA, PART OF THE DEPARTMENT OF THE ENVIRONMENT AND HERITAGE

Development of an Alert List for Alien Mammals and Reptiles

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1.0. INTRODUCTION

The terms of reference for developing an alert list of established alien mammals and reptiles are listed in Appendix 1. There is limited quantitative information on the distribution and impact of alien mammals and reptiles in Australia. The best summary of the information available for most species is contained in the Bureau of Rural Sciences (BRS) *Managing Vertebrate Pests* series, *Pest Animals in Australia* (Wilson et al, 1992), *Australia's Pest Animals – New Solutions to Old Problems* (Olsen, 1998) and in the Threat Abatement Plans for foxes, feral cats and rabbits produced by Environment Australia. All these publications acknowledge that the impact of most pests is poorly known. Except for a few species and their agricultural damage, the relationship between pest animal density and the level of damage is not known. Hence it is difficult for managers to plan and cost pest management programs because the level of control required to achieve a desired environmental outcome is not known or is difficult to estimate. Studies are required to determine the impact of pests and the relationship between pest density and damage. For most situations adaptive management experiments based on 'learning by doing' offer the best prospect for obtaining this information in a cost-effective manner (See Olsen 1998).

The species assessed in this report and their natural distribution is shown in Table 1

Table 1. List of species and their natural distribution

Animal	Natural distribution
Brown hare, *Lepus capensis*	If the European form is correctly identified with the African form (debatable by Lever), then ranges from the Savannah, steppe and semi-desert regions of Africa, all of Europe south of the coniferous forest limit and through similar habitats across Asia as far east as central China
Rabbit, *Oryctolagus cuniculus*	Iberian peninsula north to the French Pyrenees and possibly northwestern Africa.
European red fox, *Vulpes vulpes*	Eurasia (except the southeastern tropical zone), northern Africa, most of Canada and the USA.
Domestic cat, *Felis catus*. Thought to be descendant mainly from the wild cat of Africa and southwestern Asia, *F. silvestris libyca*.	Occupies a variety of forested, open and rocky country
Feral horse, *Equus caballus*	Probably once found in the wild throughout the steppe zone from Poland and Hungary to Mongolia.
Feral donkey, *Equus asinus*	Once found in the wild from Morocco to Somalia and from Mesopotamia to Oman. Inhabits broken, undulating, stoney desert country. Occupy grasslands, steppes, semi-arid shrub, and sub-alpine meadows. There are also reports that true wild horses occupied forests in Europe.

Table 1. List of species and their natural distribution (continued)

Animal	**Natural distribution**
Feral pig, *Sus scrofa*. Other species may have contributed, for example, *S. celebensis* in Indonesia.	Originally found from southern Scandinavia and Portugal to southeastern Siberia and Vietnam, from Morocco to Tunisia and in Britain, Ireland, Sri Lanka, Honshu, Taiwan, Hainan, The Malay Peninsula, the Riau Archipelago, Indonesia. Need access to free water and some vegetation cover.
Camel, *Camelus dromedarius.*	Once natural range was throughout the Arabian region (probably up to 2,000 years ago) but known with certainty only in the domestic and feral state.
Banteng, *Bos javanicus*	Originally found in Burma, Thailand, Indochina, Malay Peninsula, Java and Borneo. Tends to occupy dense thickets and forests for shelter but will occupy nearby dry and open areas. On IUCN endangered list.
Feral Asian water buffalo, *Bubalus bubalis*	Originally found from Nepal and India to Viet Nam and the Malay Peninsula, in Sri Lanka and Borneo and probably Sumatra and Java.
Indian Palm Squirrel, *Funambulus pennanti*	In the wild occurs in extreme southern Iran, Pakistan, India and Nepal. Some of the 5 related species prefer the open palm and scrub growth of the low altitudes; others frequent dense jungle and tall trees but the latter are rare. Will live around settlements and show little fear of humans.
Axis, Chital or spotted deer, *Cervus axis.*	Natural distribution is India, Nepal and Sri Lanka.
Fallow deer, *Dama (Cervus) dama.*	Originally the Mediterranean region probably including north Africa, and eastwards to southern Iran.
Red deer, *Cervus elaphus*.	Originally the palaearctic region from northern Britain to Manchuria and from south of the Arctic Circle in Norway to the Himalayas and the Tunisia/Algerian border in north Africa.
Hog deer, *Cervus porcinus.*	Natural distribution, northwestern India to Indochina (including parts of Laos, Thailand, Vietnam, and Cambodia), Bawean Island (Java) and Calamian Island (Philippines).
Timor of Rusa deer, *Cervus timorensis*	Original range is uncertain but is believed to have been widely distributed the Malaysian and Indonesia archipelagos by early Malay and other voyagers. Now mainly in Java, Clebes and the Lesser Suda Islands.

Table 1. List of species and their natural distribution (continued)

Animal	**Natural distribution**
Sambar, *Cervus unicolor*	India and Sri Lanka to southern China, Java, Borneo, Celebes and the Philippines.
Feral goat, *Capra hircus.* Descended from the wild goat or bezoar *(C. aegagrus)*.	C. aegagrus ranges from Asia Minor through the Caucasus and southern Turkmenia to Iran, Iraq and Baluchistan and western Sind and n-w India.
House mouse, *Mus domesticus*	Originally the dry steppe zone of the southern Palaearctic, and possibly the Mediterranean region.
Black rat, *Rattus rattus*	Originally probably confined to southeastern Asia.
Brown rat, *Rattus norvegicus*	Southeastern Asia south of the Himalayas.
Red-eared slider *Trachemys scripta elegans*	Freshwater systems: from Indiana to New Mexico, through Texas to the Gulf of Mexico.
Wolf snake *Lycodon capucinus*	Widely distributed through Indonesia and surrounding countries. Accidentally introduced to Christmas Island
Grass skink *Lygosoma bowringii*	Widely distributed through south-east Asia. Found only in disturbed habitats on Christmas Island where it was accidentally introduced.
House gecko, *Hemidactylus frenatus*	Widely distributed through south-east Asia. Probably accidentally introduced to northern Australia and Cocos Keeling and Christmas Island (may be naturally occurring here). In Australia, it occurs around various settlements around the northern coast of Australia from Darwin to Cairns. Seems to be dependent on humans for continued existence as it has disappeared from abandoned settlements.

2.0. Summary of distribution, impact and potential to spread

This section is a summary and review of information on naturalised non-native mammals and reptiles. Three species, feral cattle (*Bos taurus*), feral sheep (*Ovis ovis),* and wild dogs (*Canis lupis familiaris)* have not been included. The occurrence of the first two is closely associated with the contemporary stock industry. They will undoubtedly continue to escape and establish small feral populations as long as the respective stock industries remain in all the areas in which they currently occur. Also stock are run virtually free-range through much of the rangeland so that it is difficult to distinguish the impact due to domestic sheep and cattle from that caused by their feral counterparts.

Wild dogs are a sub-species of the dingo (*Canis lupis dingo)* although there are hybrids of the two. Dingoes were first introduced to Australia approximately 4,000 years ago. In most of Australia, dingoes are regarded as a native mammal that is in dynamic harmony with the natural environment (Fleming et al, IN PRESS). However, they are controlled where they, wild and domestic dogs cause extensive stock losses. While it is still debated, in core natural habitat hybrid dog packs are believed to behave similar to dingoes and be in dynamic equilibrium with their environment and hence can be regarded as 'native' (Fleming, et al, IN PRESS). Outside these core, relatively stable areas, dingoes and wild dogs cause stock damage and some loss of native animals. However, while dogs continue to be highly valued domestic pets, they will continue to range through peri-urban and agricultural areas. Other than the current dog control programs, there is little more that can be done to control the impact on native wildlife of free-ranging domestic dogs and dingoes and wild dogs that break through the buffer zones from the core areas.

Significant use has been made in the report of the climate matching assessment of established alien mammals to help estimate the potential distribution of mammals (Mary Bomford, BRS, pers. comm., 2000). Dr Bomford concluded that the extent to which the climate of the overseas distribution of a species matches significant parts of Australia is an important factor in determining the likely success of an introduced animal establishing in Australia and is also a good guide to its potential range. The maps that Dr Bomford has generated for the established introduced mammals in Australia are attached (Appendix 2) and are referred to in the following summary of the distribution and impact of introduced mammals and reptiles. Her assistance in providing this information is acknowledged and has greatly added to the value of this report. It should be recognised however, that climate matching is only a guide and other factors can influence the success and spread of a species. For example, much of Australia appears suitable for feral pigs based on climate matching but their distribution is limited by the availability of free water and suitable cover from the heat. In contrast, the distribution of rabbits is greater than might be expected from climate matching probably due to the ability of rabbits to avoid extreme heat and cold by constructing deep warrens. Some of these factors are discussed in the summary for each species.

2.1 Brown hare, *Lepus capensis*

Current and potential distribution (Figures 1a and 1b, Appendix 2)

There is some doubt about whether the Eurasian and the African hare are the same species (Lever, 1985). If they are, then hares are one of the most widely distributed of all mammals. It ranges from the Savannah, steppes and semi-desert areas of Africa, all of

Europe south of the conifer forest limit and through similar habitats across Asia as far east as central China (Lever, 1985).

In Australia they occur from Ceduna in South Australia, throughout Victoria and most of NSW and as far north as Cairns in Queensland. They favour open grazing and cropping country interspersed with some cover such as tussock grass or shrub. Climate matching assessments by Mary Bomford (BRS, pers. comm., 2000) indicate that hares have the potential to occupy a much greater range than they currently do. They can quickly take advantage of newly cleared areas but invasion seems to be limited by the availability of new plant growth, dingoes and rabbits. Hares are most common in habitats that are unsuitable to rabbits such as the black-soil plains where rabbits cannot construct extensive warrens (Wilson et al, 1992).

Impact on native species

There is little information on the impact of hares on native wildlife although they cause significant damage to pasture and to crops, especially tree seedlings.

Invasive ability and rate of spread

Hares are solitary animals, mating in winter and giving birth in spring/summer. In much of Australia, breeding appears to be seasonal. Average litter size is 2.3 with a maximum of four. In Europe, females become sexually mature at eight months and may produce up to 8 litters in a good season (Wilson et al, 1992). Jarman (1986a) states that the mean production of leverets per female per year in Australia is 8 to 10. Hares are much more mobile than rabbits but individual movements are usually less than one kilometre (Jarman 1986a). Hares are only second to rabbits and foxes in the rate at which they spread in Australia (Caughley, 1977).

2.2 Rabbit, *Oryctolagus cuniculus*

Current and potential distribution (Figures 2a and 2b, Appendix 2)

24 wild rabbits from England were released, near Geelong in 1860. Six years later, 14 253 rabbits were shot on the release site. Initially, rabbits spread slowly from Geelong and from a second release point at Kapunda in South Australia, taking about 15 years to reach the New South Wales border (Rolls 1969). Fifteen years later they were into Queensland, and by 1900 they were in Western Australia and the Northern Territory. The rate of advance varied from 10–15 kilometres a year in the wet and forested country to over 100 kilometres a year in the rangelands. It was the fastest rate of any colonising mammal anywhere in the world (Caughley 1977), although it was closely matched by the fox *(Vulpes vulpes*) and the hare *(Lepus europaeus*) in Australia (Jarman 1986a, b).

Rabbits are one of the most widely distributed and abundant mammals in Australia. South of the Tropic of Capricorn they occur almost everywhere except at the highest altitudes, in dense forests or on certain soil types such as the cracking black soil plains. North of the Tropic of Capricorn their distribution is more fragmented, they are often restricted to deep or shaded warrens on the more fertile soils in run-on areas, or to areas with a shallow watertable. Tall tropical grasslands are nutritionally inadequate for rabbits, and pasture growth occurs at the wrong time for rabbit breeding. In the more arid areas below the Tropic of Capricorn, local distributions of rabbits change dramatically with time. After a run of good seasons, rabbits may be abundant over an entire region. During severe droughts they disappear completely from some land systems and their range contracts to refuge areas where there are large, deep warrens alongside drainage channels or dried-up swamps (Williams et al , 1995). Soils are a major factor influencing local and regional distribution (Parer and Libke, 1985).

Two diseases have had a major impact on the distribution and abundance of rabbits in Australia. The first, myxomatosis, became established in the early 1950's. It spread quickly and in some places it killed 90% or more of the rabbit population. While the disease has attenuated and there has been some resistance developed to the disease by rabbits, it nevertheless still periodically causes significant reductions in rabbit density. In some areas myxomatosis has permanently eliminated rabbits. Myers 1962 showed that in parts of the Riverina district of NSW, rabbits failed to recolonise after the initial impact of myxomatosis. Myers believed that the habitat became unsuitable to rabbits due to collapse of major warren systems and growth of vegetation cover to a stage that it was no longer suitable for rabbits.

The second more recent disease was Rabbit Calicivirus Disease (RCD). It escaped from pen trials and became endemic in late 1995. Similar to myxomatosis, RCD led to a 90% or more reduction in rabbit density in parts of arid and semi-arid Australia (Neave, 1999). However, the disease has been much less effective in the cooler temperate regions. The long-term impact of RCD is unknown, but in parts of Europe, rabbit populations have recovered and remained at approximately 60% of their pre-RCD densities (Neave, 1999).

Impact on native wildlife

Rabbit damage is considered to be more severe in the rangelands, where a whole suite of plant species and their dependent animals are threatened with severe range contraction or extinction. The Threat Abatement Plan for Rabbits (EA, 1999b) lists several species that are endangered by rabbits. They include Gould's Petrel *Pterodrama leucopetra leucopetra,* the greater bilby *Macrotis lagotis,* and 17 endangered native plants including *Darwinia carne*a and *Grevillea maccutcheonii*. The effect of the rabbit in preventing regeneration of native plants is not always obvious. Many of these plants are long-lived but the populations are reaching a stage where many individuals are dying from old age. If rabbits are not controlled before the remaining plants reach the end of their reproductive lives, there will be a long-term decline of the tree and shrub populations in many parts of the rangelands.

As well as causing detrimental habitat change, rabbits threaten native mammals directly through grazing competition and possibly indirectly through intensified predation by cats and foxes after rabbit numbers crash during droughts or myxomatosis and RCD outbreaks. Unfortunately it is probable that reducing rabbit numbers will reduce numbers of native birds of prey as rabbits are the main food of many raptors during their breeding seasons.

Rabbits, in combination with other wild grazers and livestock, cause damage to the long-term sustainable use of rangeland for nature conservation and pastoralism. Rabbits cause changes in the quality of forage and damage to the flora and habitat of native fauna. Rabbit damage is most severe during and coming out of drought.

The ecological effects of rabbits on islands can be severe, but they have rarely been well documented. Philip Island, off Norfolk Island, was reduced almost to bedrock by firstly goats and rabbits and then rabbits alone. Prior to the eradication of rabbits in 1986 it was almost devoid of vegetation. The endemic parrot *(Nestor productus*) became extinct, even in the absence of introduced predators. Two endemic plant species had become extinct and one, the Philip Island hibiscus *(Hibiscus insularis*), was on the brink of extinction (Coyne, 1982). Since 1986, the island has become extensively revegetated. The Norfolk Island abutilon *(Abutilon julianae*), last seen in 1912, has recolonised patches on the island (Bridgewater and Potter, 1993).

Rabbit control can also have some unintended effects on native wildlife. Rabbit eradication on some islands has resulted in the eruption of some exotic plant species, boxthorn *(Lycium ferocissimum*) on Monunau Island (Taylor, 1968) and 'kikuyu' *(Pennisetum clandestinum*) on Bowen Island making of the islands less suitable for nesting penguins and sea birds.

Past rabbit management has also had a major impact on native fauna. On many properties there was, prior to myxomatosis, intensive poisoning and trapping of rabbits. Medium-sized mammals would have been vulnerable to traps and all grain and flesh-eating mammals and birds vulnerable to poisoning. Small rodent-size mammals would also have been affected.

Invasive ability and rate of spread

Rabbits hold the record for spreading the fastest of any introduced mammal in the world (Caughley, 1997). The rate of advance in the arid rangelands was approximately 100 kilometres a year. As in Europe, human changes to the natural environment made it more suitable for rabbits. Felled timber provided abundant rabbit harbour, and the grazing of perennial grasses by domestic stock made the grasses more nutritious and available to rabbits. The introduction of more nutritious annual grasses and forbs of Mediterranean origin, which have a seasonal growth cycle more in tune with the rabbit's breeding season, also helped. The northward spread of rabbits in Queensland in the 1900's is the result of improvements in the nutritionally poor tropical pastures and the planting of winter crops (Williams et al, 1995). The burrows of native animals such as wombats, bettongs and hare wallabies aided its spread, as did the wholesale destruction of predators such as native cats, eagles and feral cats.

They have a high reproductive rate. For example, a pair of rabbits in an outside enclosure in Canberra increased to 184 in 18 months without supplementary food (Williams et al, 1995).

2.3 European red fox, *Vulpes vulpes*

Current and potential distribution (Figures 3a and 3b, Appendix 2)

Currently, the red fox is found throughout southern half of mainland Australia. It occurs in a range of habitats from closely settled areas to arid rangelands and the alpine region. Except for Tasmania and islands such as Kangaroo Island, foxes probably occupy all the available suitable habitat, with its distribution being limited by the tropics and possibly dingoes. While foxes will enter dense, wet forest, their numbers appear to be reduced in these areas (Catling and Burt, 1995). Tracks and other access modes into dense forest may open up areas to foxes.

Impact on native species

Predation by the fox is considered to be a major threat to the survival of native Australian fauna, with non-flying mammals in the critical weight range between 35g and 5 500g and ground-nesting birds at greatest risk (Burbidge and McKenzie, 1989). Reptiles, amphibians and invertebrates are also preyed upon by the fox.

The best evidence for the damage that foxes cause to native wildlife comes from studies in Western Australia. Fox control in Western Australia not only led to increases in the populations of rare animals but to an increase in the area used and/or use of additional habitat types by these rare animals (Kinnear et al., 1988, and Saunders et al., 1995). This suggests that foxes were restricting prey animals to areas that offered cover from predation.

Fox predation has been implicated in limiting habitat choice and population size of a number of medium-sized marsupials. Even at low densities foxes can eliminate remnant populations and jeopardise species recovery programs (Short et al., 1992). Localised declines of some medium-sized mammal species, including decline in populations of Yellow-footed Rock-wallabies have been attributed to fox predation (Saunders et al., 1995). In a recent paper Short (In Press) concluded that foxes were the major cause of the decline in rat-kangaroos (Potoroidae) in NSW. He analysed bounty payments for these animals to show that the major decline in these once widespread species occurred only after the build up of foxes. Rat-kangaroos appear to have survived the impact of early extensive clearing, droughts and the introduction of rabbits and domestic stock. However, these early factors may have contributed to the decline of rat-kangaroos by making these animals more vulnerable to fox predation.

The national Threat Abatement Plan for the European Red Fox (EA, 1999c) and Scientific Committee for the NSW *Threatened Species Conservation Act 1995* have identified the European Red Fox as threatening numerous Endangered and Vulnerable species including the Malleefowl *Leipoa ocellata*, Hastings River Mouse *Pseudomys oralis*, Mountain Pygmy-possum *Burramys parvus*, Broad-toothed Rat *Mastacomys fuscus*, Long-footed Potoroo *Potorous longipes*, Little Tern *Sterna albifrons*, Yellow-footed Rock-wallaby *Petrogale xanthopus*, Brush-tailed Rock-wallaby *Petrogale penicillata*, and Southern Brown Bandicoot *Isoodon obesulus*. While all these species fit within the critical weight range for those animals that are believed to be threatened by foxes (Burbidge and McKenzie, 1989), there have been few good studies that clearly demonstrate the threatening impact of foxes on native fauna in NSW. Priddel and Wheeler (1990, 1997) showed that foxes were a primary cause of predation to Malleefowl, however, foxes were not the only cause. Stock, feral goats, birds of prey, fire and the availability of suitable seed for chicks were other major factors influencing the survival of Malleefowl (Priddel and Wheeler, 1990 and 1997).

Although there have been many observations of fox predation on native animals and the occurrence recorded of native fauna in the diet of foxes, these in themselves does not necessarily imply that foxes are the cause or a major cause in the decline of a species. Under normal circumstances, most species produce many more offspring that is required to maintain or increase their population. It is possible to take a substantial ongoing harvest of animals from a population without causing a decline in numbers (Bomford et al., 1995). Predation *per se* is not the issue but the impact it has on recruitment to the next generation and to the overall stability of the population. However, population studies linking the impact of fox predation to the density of prey species are difficult to conduct and rarely done.

It is important to appreciate the complexity in identifying fox damage to wildlife. Foxes are only one factor affecting the long-term conservation of native animals. Other factors including changed fire regimes, other pests such as rabbits and feral goats, loss of habitat and habitat fragmentation have undoubtedly also been very important. The latter two factors have also probably given foxes better access to native shrub habitat and to the native animals it contains. Fox control alone will rarely ensure the survival of threatened and endangered fauna that are being predated by foxes.

Prey switching

Because rabbits are the primary prey of foxes through most of their range, there is concern that foxes and other predators will switch to or prey more heavily on native animals if rabbit density falls rapidly due to control operations or disease outbreak such as RCD. To date there is no good information on this issue although studies are being conducted under the RCD Program to try and answer it. One theory suggests that if prey switching is a problem, the damage from a rapid fall in rabbit numbers may have already occurred. Crashes in rabbit numbers have been occurring for many years as rabbit densities crash due to drought and other factors such as myxomatosis outbreaks. Others feel that a drop in rabbit numbers will lead, after a time lag, to a significant fall in fox density and therefore be of overall benefit (Williams et al., 1995). Until good information is available, the RCD Program recommended that in those areas with significant rabbit and predator populations as well as significant populations of endangered and threatened species that are susceptible to predation, predator control should be considered following outbreaks of RCD (Neave, 1999).

A subset of this issue is the need to control feral cats along with foxes. The concern is that feral cats will increase and prey more heavily on native fauna once foxes are removed. Risbey and Calver (In Press) obtained evidence for an increase in feral cat numbers following fox control in Western Australia. They suggest that the cats prey more heavily on small native mammals and concluded that the increase in feral cat numbers after fox control alone can have a greater impact than no fox or cat control. However, Molsher (1998) observed no significant increase in feral cat numbers or predation rates following fox control in the Central Tablelands of NSW although she showed that feral cats increased their use of forest habitats. While control of feral cats along with foxes might be an advisable precaution where some rare small mammals are at risk, at present this is not practicable due to the lack of effective techniques for controlling feral cats.

Impact on non-target animals due to fox control.

Several non-target animals are at risk from 1080 poisoning for foxes. They include farm dogs, and native animals such as tiger quolls. Recent work on extent of caching baits by foxes for later use (Kay et al., 1998) highlight the need to take special care over the use of poison baits. They found that foxes can cache about 10% of buried 1080 baits for later use, leaving them in a position where they are likely to be more available to non-target wildlife. The risk to non-target animals needs to be carefully assessed before baiting. The risk of non-target losses can be reduced by strictly following recommended baiting procedures including using the appropriate bait and quantity of poison and, where appropriate, by burying the bait. Free-feeding with non-poisoned bait and checking the tracks around the baits can help determine whether non-target animals are at risk.

Invasive ability and rate of spread

The fox in Australia showed the fastest rate of spread of any introduced mammal in the world except for the rabbit in Australia (Jarman, 1986b). The earlier establishment of rabbits undoubtedly assisted the spread of foxes mainly by providing an abundant and readily available food source. The fox crossed from Victoria into New South Wales 13 years after the rabbit and entered Queensland from the New England district at the same time as the rabbit; it entered Queensland in the far west 24 years after the rabbit.

Foxes usually live in family groups with only limited overlap of their home ranges (Saunders et al., 1995). Home range size in Australia varies from 30 hectares in some urban areas to about 500 hectares in alpine areas. The size seems to depend on the availability of food and other important resources such as den sites. Adult foxes rarely travel more than 10 kilometres in a day although under exceptional circumstances, dispersing foxes may move 100 kilometres or more in search of a new territory (Saunders NSW Agriculture, pers. comm. 1998). Typically these great dispersers are young male foxes, usually in late summer through to the start of breeding in winter.

2.4 Domestic cat, *Felis catus*.

Current and potential distribution (Figures 4a and 4b, Appendix 2)

There is debate about when cats first arrived in Australia but it likely that it preceded European settlement. They were common companions on the ships of early explorers and it is likely that some escaped or survived shipwrecks off the coast of Australia as far back as the 1600's. Cats were deliberately released during the 1800's to control the burgeoning rabbit numbers, a tactic which like releases of other predators such as mongoose, failed to control the spread and increase in rabbits across central and southern Australia.

Feral cats are found throughout Australia including several off-shore islands. Their close association with humans has greatly assisted their spread and continued existence in a wide range of habitats. Also they are highly adaptable being able to survive and reproduce from the tropical north to Macquarie Island in the south.

Impact on native species

It is well documented that feral cats take and kill a range of native animals including small mammals, birds and reptiles (Wilson et al, 1972, EA, 1999a). There is also good evidence that they have been responsible for the massive decline if not extinction of native fauna from several off-shore islands of Australia and elsewhere (Burbidge, 1989; Copley, 1991). Feral cats frequently are infected with the protozoan parasite, *Taxoplasma gondii*, which can be transmitted to domestic and native animals through cat feaces. The disease, taxoplasmosis can cause infertility, blindness and even lead to death of native wildlife (David Spratt, CSIRO Wildlife and Ecology, pers. comm., 1986). However, available quantified information on the damage that cats cause generally to native wildlife is poor. Much of it is anecdotal and observation, not based on sound experimental design. Certainly they have been shown to have significant impact on the survival of small mammals such as Marla or rufous hare-wallaby (*Lagorchestes hirsutus)* in reintroduction programs. But other factors such as poor predator sense of the captive reared wildlife and insufficient or poor habitat for the reintroduced species may have greatly contributed to the lack of success of these re-introductions. Nevertheless, available evidence about the impact of feral cats is sufficient to consider this species a major threat to several species of native fauna (EA, 1999a). Nationally endangered species that are known or perceived to be under threat from feral cats are listed in the Threat Abatement Plan for Predation by Feral Cats (EA, 1999a). Species include the eastern barred bandicoot (*Perameles gunnii),* numbat (*Myrmecobius fasciatus)*, rufous hare-wallaby and the little tern (*Sterna albifrons).*

Invasive ability and rate of spread

Cats are non-seasonal breeders. When sexually mature (approximately 10-12 months), females can have two litters a year, producing on average, 4 young per year. However, there is a low survival of the young during their first year, mainly due to insufficient food in winter (Jones and Coman, 1982), resulting in a fairly stable population density throughout their range. It is likely that feral cats occupy most of the habitat currently available to them in Australia. They are most abundant in peri-urban and rural areas, with their density being much lower in natural or near natural forest. Hence, maintaining the integrity and improving the quality of remaining forest is likely to reduce its suitability to feral cats as well as providing better protection and increased resources to wildlife.

2.5 Feral horse, *Equus caballus*

Current and potential distribution (Figures 5a and 5b, Appendix 2)

Horses were first introduced to Australia with the First Fleet in 1788. There numbers rapidly increased to approximately 160,000 by 1850. The first record of horses escaping into the wild or being abandoned was 1804 (Rolls, 1969). Infrequent musters and inadequate fencing led to the escape of more horses and the growth of feral herds. Feral horses were considered a pest by the 1860's (Rolls, 1969).

Australia has the largest number of feral horses in the world, estimated to be 300,000 in 1993 (Dobbie et al, 1993). They occupy most of the habitat to which they are suited in Australia. However, extensive management programs in recent years have probably reduced that number significantly. Feral horses are widely distributed and most common throughout most of the cattle-raising districts of Queensland, the Northern Territory and to a lesser extent, Western Australia. However, they also occupy several small parks and state forests in southern Australia with significant populations totaling several hundreds of individuals in the southern Highlands (Michelle Walter, University of Canberra, pers. comm., 2000). Feral horse populations fluctuate significantly in response to seasonal conditions, and human intervention. Their numbers increase in wetter seasons and decline during dry cycles.

Impact on native species

The impact of feral horses has only been studied in Central Australia and, to a lesser extent, in the southern highlands. However, although there is good observational evidence for the environmental damage due to horses, it has not been well quantified (Wilson et al, 1992; Dobbie et al, 1993). The damage due to horses is often confounded by other factors such as grazing by stock and other feral and wild grazers and changes in the long-practised burning regimes of traditional owners. Nevertheless, feral horses are believed to threaten a range of native wildlife including several threatened species, especially during and coming out of drought when they tend to concentrate around and severely pressure vegetation around remaining water points. In Central Australia, they can foul water holes with their carcases, accelerate gully and sheet erosion and denude large areas of vegetation, forcing macropods from their favoured habitat. Dyring (1990) found that areas frequented by feral horses in the southern highlands had fewer native plants and contained more weed species. She also suggested that trampling by feral horses caused erosion, changes in hydrology and siltation.

Invasive ability and rate of spread

Feral horses usually form small social units called harem and bachelor groups. Harem groups usually contain a dominant stallion with several mares and their offspring. Besides human control, the primary cause of death is associated with drought through lack of water, starvation, and consumption of usually avoided toxic plants. Under favourable conditions, feral horse populations can increase by 20% a year (Dobbie et al, 1993).

2.6 Feral donkey, *Equus asinus*

Current and potential distribution (Figures 6a and 6b, Appendix 2)

Donkeys now have an almost global domestic distribution. They were used for haulage in Australia in the 1850's through to the early 1900's. Following the introduction of motorised transport in the early 1900's, domesticated donkeys were liberated and they built up large feral populations.

Donkeys have been successful colonists in Australia for several reasons. They can tolerate the heat and aridity of semi-arid regions; are well adapted to life in barren and inhospitable terrain; they can feed on a wide variety of plant material; and can dig up to 10 cm for water. Feral donkeys are most abundant in the Kimberley region of Western Australia and the Victoria River District of the Northern Territory mainly between latitudes 15^0 and 20^0 south. They are also relatively common in arid central Australia, the Hamersley Range of Western Australia and in pockets in Queensland, NSW and South Australia (Wilson et al, 1992). Climate matching suggests that there are large sections of inland Australia that they could occupy. Their expansion these areas is probably limited by human control and fencing.

Impact on native species

Like most pest species, the damage due to donkeys has not been well quantified. It is likely however, that it is similar to that due to feral horses which occur in similar habitats through much of Australia.

Invasive ability and rate of spread

Donkeys have been studied in the Northern Australia by (Choquenot, 1989) and (Wheeler, 1987). Choquenot estimated the annual rate of increase for parts of northern Australia for a recovering population to be 0.2 or 20% per year. For example, an uncontrolled population in the McArthur River area of the Northern Territory increased from 40 in 1936 to 1,500 in 1966 (Lever, 1985).

2.7 Feral pig, *Sus scrofa.*

Current and potential distribution (Figures 7a and 7b, Appendix 2)

Feral pigs are widely distributed in Queensland, the Northern Territory, New South Wales and the Australian Capital Territory (Wilson et al 1992, Choquenot et al, 1996). There are isolated populations in Victoria, Kangaroo Island in South Australia and in Tasmania. Their distribution in western Queensland and NSW is closely related to the location of inland watercourses and their associated flood plains (Choquenot et al, 1996). The spread of feral pigs along the inland water courses of NSW and Queensland is believed to have occurred approximately during the last 40 to 80 years (Choquenot et al, 1996). There are estimated to be between 3.5 million and 23.5 million feral pigs in Australia inhabiting approximately 38% of mainland Australia (Hone, 1990). However, their distribution and abundance can vary considerably, depending on environmental conditions. Numbers fall dramatically during extended dry periods and human control programs, but feral pigs can rapidly recover their numbers during a run of good seasons.

Feral pigs are relatively intolerant to heat. Hence their distribution is largely limited by the lack of cover and access to free water. However, climate matching indicates that there are extensive areas that feral pigs could occupy where they are currently absent or in low density (Mary Bomford, BRS, pers. comm., 2000). These include large parts of central and eastern Tasmania, Eyre Peninsula and the south-east of South Australia, and south-western Western Australia.

Impact on native species

Feral pigs are believed to cause serious damage to native wildlife, however, there is little good quantified information to support this (Choquenot et al, 1996). They are believed to degrade native habitat through selective feeding on native vegetation, trampling and rooting soft areas as well as predation on and competition with native animals. For example, they have been recorded damaging the nests of green turtles and eating the eggs although the overall impact of this damage is difficult to quantify given the intrinsic very high mortality of turtle hatchlings (Choquentot et al, 1996). While the rooting of pigs can be obvious, Mitchell (1993) found that in the Queensland Wet Tropics, feral pigs only rooted 4.3% of the ground surface. Although not proven, there is believed to be a strong correlation between rooting damage by feral pigs and soil moisture, soil friability and probably the presence of large numbers of soil invertebrates and bulb producing plants.

There is also concern that the rooting habit of feral pigs may help to spread the soil fungus *Phytophthora cinnamoni* but again this risk is not well quantified.

There are potential indirect consequences due to the presence of feral pigs. Similar to that for many other exotic animals, poisoning with 1080 baits can threaten non-target native fauna such as dingoes and tiger quolls. Also, there is a concern that the dogs that feral pig hunters use in the wet tropics can take non-target wildlife such as the chicks of the endangered cassowary *(Casuarius casuarius*) as it is not possible for hunters to continually control their dogs during hunting forays.

Invasive ability and rate of spread

As mentioned above, the distribution of feral pigs is largely determined by their intolerance to heat and limited by their access to free water and dense cover. Their distribution contracts as the large inland water areas of eastern Australia dry but it rapidly expands when the waterways fill again.

Feral pigs are generally gregarious with the basic social group consisting of one or more sows and their piglets. Other social groups consist of young females, young males or some mixed combinations. However, adult boars are usually solitary. Group size can vary from a few to 50 individuals but mobs of 100 or more may form around remaining water sources during dry periods (Choquenot et al, 1996). The size of their home range and movements are largely determined by the distribution and abundance of food and other essential resources such as water and cover. For example, in the Wet Tropics of north Queensland, feral pigs move to the coastal plains during the seasonal dry to feed on sugar cane and tropical fruits. They tend to move back to the wet tropic rainforest when the seasonal wet begins (McIlroy, 1993). Similar seasonal movements occur in Northern Territory feral pigs in response to the tropical wet dry cycle.

The reproductive potential of pigs is closer to that of the rabbit than other large mammals in Australia. Under favourable conditions, adult sows can produce two weaned litters in 12 – 15 months with an average litter size of 5-6 (Choquenot et al, 1996). In tropical seasonal habitats, sows usually produce only one litter per year. Females become sexually mature at approximately 25kg, a weight they usually reach at 7-12 months.

2.8 Camel, *Camelus dromedarius*.

Current and potential distribution (Figures 8a and 8b, Appendix 2)

As the common name suggests, the Arabian camel was originally confined to Arabia. Australia may now have the largest wild population of camels. They occupy most of Australia's desert country including the Great Sandy, Gibson, Great Victoria and Simpson deserts, as well as much of the semi-desert lands. Camels were first introduced into Australia in the 1840's to assist in the exploration of inland Australia. Between 1840 and 1907, between 10,000 and 20,000 camels were imported from India with an estimated 50-65% landed in South Australia (Rolls, 1969). The date of the first establishment of feral camels is unknown but some escaped during the Burke and Wills expedition in 1860. The feral animal population increased substantially after the 1920's with the total population of 43,000 or more today (Nowak 1999). Climate matching (Mary Bomford pers. comm. BRS, 2000) shows that there is significant additional country that camels could occupy. Their spread however, is probably inhibited by human interference, especially as their value as a resource has been recognised (Dave Wurst, NT Parks and Conservation Commission, pers. comm., 1999).

Impact on native species

There is little data in the damage that camels cause to native wildlife. Like other wild feral and native herbivores, they undoubtedly add to the total grazing impact with the damage likely to be most severe during and coming out of drought. However, given their adaptation to desert conditions, camels are likely to cause less impact than those species such as feral horses, feral donkeys and domestic stock which are more concentrated around water points during extended dry periods.

Invasive ability and rate of spread

Camels form a range of groups from bachelor groups, which young males join after their second year, adult females and their newborn and family groups containing up to 30 adult females along with their one and two year old offspring (Nowak, 1999). Feral aggregations of up to 500 individuals have been recorded in Australia (Nowak, 1999). Like feral horses, female camels usually give birth to one young every second year. Hence they do not have a high potential rate of increase but this is compensated to some extent by their longevity and few morality factors, other than by human intervention.

2.9 Banteng, *Bos javanicus*

Current and potential distribution (Figures 9a and 9b, Appendix 2)

Bantang originally occurred naturally in the Indo-Malaysian area including Borneo, Malaysia, Java and Burma. They are now restricted to a few isolated herds on Java and in south-eastern Asia. They are the ancestor of Bali cattle which were bred on Bali, Timor, Sulawesi, and Borneo (Lever, 1985). Bantang are listed as endangered in Indonesia by the IUCN and the Australian herd on the Coburg Peninsula is considered to be one of the largest in the world (Nowak, 1999; Wilson et al, 1992). The strain of Bantang imported to Australia between 1829 and 1940 was the domestic strain. They were originally held at Port Essington but established wild populations when the settlements on the Coburg Peninsula were abandoned (Lever, 1985). By 1964, Bantang ranged over the whole 1,800 square kilometres of the Peninsula, and occasionally moved further south. They are mainly associated with freshwater swamps and lagoons near the coast. The population has been culled from time to time but still exists through much of the Peninsula where approximately 1,000 still exist (Nowak, 1999).

Bantang are well adapted to the freshwater wetlands and poor dry-season pasture of northern Australia. Climate matching of Australian conditions with their natural range suggest that unchecked, Bantang could inhabit large sections of wet-dry tropics of northern Australia (Mary Bomford, BRS, pers. comm., 2000).

Impact on native species

The impact of Bantang has not been studied but they are reported to trample and overgraze the sandy plains of the Coburg Peninsula. The common sedge (*Fimbristylis cymosa)* seems to be highly favoured (Lever, 1985).

Invasive ability and rate of spread

There is little information on the invasive ability and rate of spread of Bantang although it likely to be similar to that of water buffalo. They become sexually mature at about two years and can produce one or two offspring per year. They occur in groups of from 2 to 40 individuals (Nowak, 1999).

2.10 Feral Asian water buffalo, *Bubalus bubalis*

Current and potential distribution (Figures 10a and 10b, Appendix 2)

Buffalo were introduced to northern Australia between 1826 and 1866. They rapidly became feral and occupied all major habitats in the top end of the Northern Territory above 16^{o}S (Skeat, 1990). In 1985-86, Baylis and Yeomans (1989) estimated that there were 350,000 feral buffalo in the Northern Territory. Following an intensive program to reduce wild reservoirs of brucellosis and tuberculosis as part of the BTEC Program, buffalo are now restricted to a few areas in the Northern Territory, primarily in Arnhem Land but also in low density in the wetlands to the west of the Gulf of Carpentaria. However, they have the potential to reinvade the area that they previously occupied. Should management be relaxed, they have the potential to increase substantially in density and re-occupy large parts of the Northern Territory and also Northern Queensland and Western Australia. This potential is supported by the climate matching assessment conducted by Mary Bomford (pers. comm. BRS, 2000).

Impact on native species

While not well quantified, buffalo can cause substantial environmental impact. The damage includes grazing and trampling of wetlands at times leading to changes in hydrology with the subsequent loss of swamps and billabongs. Buffalo have also been reported to damage barrages in northern rivers leading to salt water intrusion and substantial changes to the fauna and flora of the freshwater systems affected. Braithwaite et al, 1984 recorded reduction in vegetation, abundance of vertebrate fauna and loss of soil nutrients and litter in monsoonal forests. In drier habitats, reduced recruitment of some native tree seedlings have been recorded.

Species at risk include nest sites of salt water crocodiles. These sites are believed to support a range of native species which are also at threat due to the impact of buffalo on crocodiles. Georges and Kennett (1988, 1989) reported that they trample the nesting ground of the relatively rare pig-nosed turtle *Carettochelys inscuplta.*

Invasive ability and rate of spread

Skeat, 1990 estimated the exponential rate of increase for buffalo in Kakadu National Park when populations had been previously reduced to low densities to be about 0.23 or 23% per year. However, he found it to be highly variable and dependent on the quality of season. He predicted positive rates of increase in 72% of years with a mean rate of increase of 10% per year. Maternal groups are loosely aggregated into herds of 30 to 500 individuals. In her 20 year lifespan, an adult female may produce up to 12 offspring.

2.11 Indian Palm Squirrel, *Funambulus pennanti*

Current and potential distribution (Figures 11a and 11b, Appendix 2)

The natural distribution of the Indian palm squirrel is the extreme south-eastern portion of Iran, Pakistan, India and Nepal (Nowak, 1999). They are commonly associated with people in towns and cities. In Australia, they are restricted to a small population surrounding Perth Zoological Gardens although some animals may have spread into few surrounding suburbs (Wilson et al, 1992). A population also escaped and established around Taronga Zoological Gardens in Sydney, but has since been eradicated (Wilson et al, 1992).

Comparisons of the climate in their natural range and in Australia, suggest that the natural habitat where they currently occur in south-Western Australia is not highly suitable to this species. Potentially, they could occupy suitable habitat in northern sub-tropical Australia (Mary Bomford BRS, pers. comm., 2000).

Impact on native species

There is no information on the pest potential of this species.

Invasive ability and rate of spread

Like many small rodents, *F. pennantii* has a high reproductive potential (Nowak, 1999). They become sexually mature at 6-11 months and can breed through most of the year although in India, they have two to three breeding peaks, with an individual female being capable of breeding three times per year. The average litter size is three. The species is gregarious and forms groups of up to 10 individuals.

2.12 Axis, chital or spotted deer, *Cervus axis*.

Current and potential distribution (Figures 12a and 12b, Appendix 2)

Chital deer occur naturally in India, Nepal and Sri Lanka (Nowak, 1999). They have formed naturalised populations in several countries including Russia, USA, Argentina, Brazil, Uraguay, Hawaii as well as in Australia.

They were first introduced to Australia in 1803, if not earlier (Lever, 1985). A herd of 400 was established at Parramatta from where a number escaped to the surrounding bush. Other populations were established in Victoria, Queensland and Tasmania (where they did not survive for long). They are not common anywhere in Australia with small populations occurring around Charters Towers, the Gulf Country and parts of western Queensland (Wilson et al, 1992).

They prefer woodland habitat and lie up during the day in riverine thickets (Lever, 1985). Comparisons of the climate in their natural habitat with that of Australia indicate that they could occupy vast areas of suitable habitat in Australia (Mary Bomford, BRS, pers comm., 2000). Although it is also suggested that their slow spread and poor survival has been variously attributed to dingo predation, the impact of droughts and the poor quality of Australian vegetation.

Impact on native species

There is little information on their impact on native wildlife, but given their current low density, it is likely to be minimal.

Invasive ability and rate of spread

Chital deer have mainly been spread and encouraged by people in other countries where they have been introduced. They can increase their numbers significantly. For example, a population in one part of Texas was estimated to be 2,200 in 1964. By 1966 it had increased to 6,450 and by 1974 to 19,581 (Lever, 1985). In Hawaii, a herd of 8 animals increased to 1,000 within 20 years. Ten years later they were believed to have increased to between 6,000 and 7,000 (Lever, 1985). However, in Australia, the major remaining population of chital deer near Charters Towers has not spread far from the original release site in 1886 (Strahan, 1983).

2.13 Fallow deer, *Dama (Cervus) dama*.

Current and potential distribution (Figures 13a and 13b, Appendix 2)

Fallow deer occur naturally in the Mediterranean region eastwards to southern Iran (Lever, 1985). They have since become naturalised in several countries and now occur in the wild in 38 countries in all six continents where they have adapted to a wide range of environmental climatic conditions. Apart from rats mice and feral domestic animals, fallow deer are the world's most widely naturalised animal (Lever, 1985). Countries where they have become naturalised include Britain, many parts of continental Europe, Sweden, Russia, South Africa, North and South America as well as Australia and New Zealand. Fallow deer were introduced to several parts of Australia and now occupy open lowland woodlands in parts of New South Wales, Queensland, Victoria South Australia although they are most numerous in Tasmania. They do not readily penetrate the densely forested mountain regions of Australia.

There are estimated to be approximately 10,000 fallow deer in Tasmania occupying an area of approximately 400,000 hectares in the east and central midlands (Wilson et al, 1992).

Comparative climate matching of their natural range with potential suitable areas in Australia indicate that fallow deer could occupy significant parts of south-western Victoria, southern South Australia and south-west Western Australia (Mary Bomford, BRS, pers. comm., 2000).

Impact on native species

Fallow deer graze a wide range of grasses and herbs and shrubs including banksia and wattles. However, their impact on native wildlife has not been quantified.

Invasive ability and rate of spread

Fallow deer are gregarious, seasonal breeders. The mating season is mid-Autumn with calves, one but rarely two per female, born in spring to early summer (Strahan, 1983; Nowak, 1999).

2.14 Red deer, *Cervus elaphus*.

Current and potential distribution (Figures 14a and 14b, Appendix 2)

The natural distribution of the red deer is the palaearctic region from northern Britain to Manchuria and from south of the Arctic Circle to the Himalayas and the Tunisian/Algerian border in North Africa (Lever, 1985). This species has formed naturalised populations in Morocco, South Africa, Argentina, Chile, Peru and in Australia and New Zealand. In Australia, red deer are moderately common in the headwaters of the Brisbane River in Queensland and in the Grampian Ranges of Victoria. A small population exists near the headwaters of the Snowy River (Wilson et al, 1992).

Red deer inhabit open undulating pastoral country interspersed with numerous water courses, to steeply wooded hills with thick vine cover. They especially favour mixed grassland, woodland and rainforest associations (Lever ,1985). Climatic assessments based on their natural range indicate that they have significant potential to inhabit a significantly greater area in Australia, especially in south-west Western Australia, Tasmania and southern Victoria (Mary Bomford, pers. comm., BRS, 2000).

Impact on native species

There is little known about their impact on the environment.

Invasive ability and rate of spread

Red deer are gregarious and seasonal breeders with mating occurring mainly during April with young born from late November to December (Strahan, 1983). In Texas, red deer on mainly unfenced land rose from 21 in 1964 to 95 on 15 ranches in 1966, 307 in 1971 and 404 in 1979 (Lever, 1985).

2.15 Hog deer, *Cervus porcinus*.

Current and potential distribution (Figures 15a and 15b, Appendix 2)

Hog deer occur naturally in north-western India to Indochina and on Java and in the Philippines (Lever, 1985). Naturalised populations occurred in Sri Lanka and Australia. Currently, hog deer only occur in isolated populations in the south-east part of coastal Victoria from Gippsland to Orbost. It is the only wild population outside the Indian sub-continent (Lever, 1985).

Climatic assessment by Mary Bomford (BRS, pers. comm., 2000) indicates that it could also inhabit significant parts of tropical north Australia.

Impact on native species

There is no information on the impact of hog deer on native wildlife.

Invasive ability and rate of spread

In common with many other tropical species, hog deer breed throughout the year although in Australia, the calving peak occurs in late winter and early spring. Usually only one young, rarely two, are produced per female per year (Strahan, 1983)

2.16 Timor or Rusa deer, *Cervus timorensis*

Current and potential distribution (Figures 16a and 16b, Appendix 2)

The original range of Rusa deer is uncertain. It is believed to have been widely distributed throughout Malaysia and Indonesia by humans (Lever, 1985). Naturalised populations occur in Mauritius, Australia, New Zealand, Papua New Guinea and in Indonesia. In Australia they occur in Royal National Park (NSW), on Prince of Wales, Possession and

Friday Islands in the Torres Strait and on North-East Island adjacent to Groote Eylandt in the Northern Territory (Wilson et al, 1992). There is additional potentially suitable climatic areas for Rusa deer in central and eastern Tasmania and around Darwin in the Northern Territory (Mary Bomford, BRS, pers. comm., 2000).

Impact on native species

There is no reliable information on the impact of Rusa deer on native wildlife although NSW National Parks and Wildlife Service is sufficiently concerned about the damage that the approximately 200 animals in Royal National Park are causing that the Service is considering culling the population.

Invasive ability and rate of spread

Rusa are gregarious with a non-distinct breeding season although in Australia, mating peak occurs during July/August with a calving peak around March to April (Strahan, 1983).

2.17 Sambar, *Cervus unicolor*

Current and potential distribution (Figures 17a and 17b, Appendix 2)

Sambar occur naturally in India and Sri Lanka to southern China, Java, Borneo and the Philippines (Lever, 1985). Naturalised populations occur in the USA, Australia and New Zealand. Sambar established from several releases in eastern Victoria and it is now continuously distributed throughout Gippsland, the Victorian Alpine district and the southern highlands of NSW. It also occurs in monsoon rainforest on the Coburg Peninsula of the Northern Territory (Wilson et al,1992). They appear to be continuing to spread north along the moist sclerophyll forests and ravines and Great Dividing Range.

Impact on native species

The impact of Sambar deer on native wildlife is not known, but they occupy and are continuing to spread into relatively fragile montaine habitat. An assessment of their impact appears warranted although it may be difficult to differentiate the damage due to Sambar from that caused by feral goats, rabbits and native grazers.

Invasive ability and rate of spread

Sambar are a solitary, tropical species with an ill-defined breeding season. However, there are breeding peaks in May/June and from September to November.

2.18 Feral goat, *Capra hircus*.

Current and potential distribution (Figures 18a and 18b, Appendix 2)

Australia’s feral goats occupy about 1.21 million square kilometres, mostly in the semi-arid and arid lands used for pastoral farming of sheep. Domestic goats occur on all continents except Antarctica, but feral populations are only common in Australia, New Zealand, and on many small islands.

In Australia, feral goats occur in all states and in the Australian Capital Territory, but are rare or absent on the mainland of the Northern Territory (Parkes et al, 1996). In 1993 there were about 2.6 million feral goats in Australia but this number has fluctuated widely. Most feral goats inhabit the semi-arid pastoral areas used for sheep farming. The most extensive populations live in semi-arid pastoral areas of Queensland, New South Wales, South Australia, and Western Australia where people, through supply of water and controlling predators to improve sheep production, have modified the natural habitat favourably for feral goats. Isolated populations of feral goats occur in the higher rainfall and agricultural areas in Victoria, Tasmania, eastern New South Wales, Queensland, South Australia, and south-west Western Australia. These goats survive mainly in areas where patches of scrub or forest offer protection from control by people. From time to time, populations are lost due to drought and human intervention and new populations established. Recolonisation of such areas cleared of goats is usually via escaping or deliberately released domestic animals rather than by dispersing feral goats. Feral goats also occur on many Australian offshore islands (Parkes et al, 1996). These include islands with important conservation values, such as Lord Howe Island and islands in the Recherche Archipelago of Western Australia. Island populations are generally considered to be pests. The establishment of new island populations is less likely now than in the past, due to the awareness of the damage that they can cause.

Based on assessment of climate areas in Australia that coincide with that of the natural range of the species; feral goats could potentially inhabit vast tracts of Australia (Mary Bomford, BRS, pers. comm., 2000). Spread to these areas is probably restricted by the presence of wild dogs north of the Wild Dog Fence.

Impact on native species

Feral goats cause an unknown, but usually assumed to be substantial loss to conservation values. Feral goats have been responsible for severe or even catastrophic environmental damage on island habitats that evolved without browsing mammals. On mainland Australia there are no documented examples of feral goats severely damaging large areas in the absence of significant populations of other herbivores, such as sheep, cattle, rabbits and kangaroos. But feral goats contribute to the damage to vegetation, soils, and native fauna in the large areas of pastoral land that are overgrazed, although their share is generally less than that of other herbivores. Feral goats do, however, have the capacity to reach high densities and inflict severe damage if left uncontrolled. Feral goats also compete with native animals for resources. They can deplete the soil's protective cover of vegetation and break up the soil crust with their hooves. They also affect trees and shrubs by eating established plants and by preventing regeneration of seedlings. Feral goats also overgraze grasses and herbs when alternative food is scarce. These impacts undoubtedly affect ecosystem processes, although the extent of the role of feral goats among all the other agents of change is difficult to quantify and may differ during droughts and wet periods. In the *Endangered Species Protection Act* 1992, the Commonwealth Government has listed '*competition and land degradation by feral goats*' as a 'Key Threatening Process' to the survival of native species.

Invasive ability and rate of spread

Feral goats were selected through the process of domestication to have characteristics of value to people. However, many of these characters are the traits that make feral goats pests. Harvested populations of feral goats can increase by over 50% per annum if harvesting stops, because goats become sexually mature and can breed in their first year although they do not reach their maximum reproductive rate until 21 months. Feral goats in arid areas centre their movements about permanent water and have much larger, non-exclusive home ranges than the more temperate parts of their range. In the goldfields region of Western Australia, adult female, sub-adult female, adult male, and sub-adult male goats had average home ranges of 69, 63, 247 and 379 square kilometres respectively (Parkes et al, 1996). The maximum individual home range was 600 square kilometres. Ranges were smaller in drier periods, presumably because the goats had to visit water more frequently.

2.19 House mouse, *Mus domesticus*

Current and potential distribution (Figures 19a and 19b, Appendix 2)

House mice probably arrived in Australia at the start of European settlement. They are now widespread throughout Australia including on many off-shore islands. They are most common in settled and disturbed areas, periodically erupting into plagues in cereal cropping areas of southern Queensland, NSW, Victoria and South Australia. They occupy most of the available habitat so that their distribution is unlikely to increase significantly in the future unless there is significant disturbance to natural habitat. House mice are relatively uncommon in natural habitats except after disturbances such as fire or clearing (Singleton and Redhead, 1989).

Impact on native species

The impact of house mice on native wildlife is not well understood or documented (Caughley et al, 1998). They may denude native vegetation during plagues and may lead to short-term impact on native wildlife by predators at the end of a plague due to prey switching as the mouse numbers decline. Also, large numbers of native raptors and owls may die from starvation as the mouse numbers dry up. However, this has not been clearly demonstrated. Probably the most serious impact that mice cause is the loss of non-target wildlife due to the use of rodenticides for mouse control (Caughley et al, 1998). Large quantities of legal and illegal poisons are used during mouse plagues. While governments endeavour to ensure that poisons are applied in such a manner that non-target losses are minimised, nevertheless, some losses of native fauna are inevitable. This may be through direct consumption of the poison bait or consumption of poisoned mice. During the 1993 mouse plague, several species of native birds died from the direct consumption of strychnine bait. They included red-rumped parrots (*Psephotus haematonotus*), bluebonnets (*Psephotus haematogaster)*, and crested pigeons (*Ocyphaps lophotes,)* (Caughley et al, 1998). Strychnine is now being phased out and replaced with zinc phosphide. The extent of non-target kills from use of this new rodenticide is not yet known but considered to be much less than for strychnine (Parker and Hannan-Jones, 1996).

Invasive ability and rate of spread

House mice can bred throughout the year, but tend to have a breeding peak between spring and autumn in most of their range. The litter size can vary from 1 to ten. Mice move to find food, water and shelter and to find breeding partners. Their home range during the breeding season is about 0.035 hectares for males and 0.015 for females. During the non-breeding season this increases to about 0.2 hectares for both sexes (Caughley et al, 1998). Mice disperse from sites where there is good cover and food to other areas where food and other resources are seasonally available. They associate closely with people and can readily be transported with goods.

2.20 Black rat, *Rattus rattus*

Current and potential distribution (Figures 20a and 20b, Appendix 2)

Like house mice, black rats probably arrived in Australia at the start of European settlement. They are now found throughout the temperate and tropical parts of Australia including off-shore islands, but are relatively uncommon in the more arid central areas. Similar to house mice, they are usually closely associated with human settlements or in highly modified habitats. They rarely occur in natural habitats (Caughley et al, 1998). Black rats probably occupy the suitable, available habitat in Australia. Their range is unlikely to change unless there is significant disturbance to remaining areas of natural habitat.

Impact on native species

Black rats are significant predators of reptiles and birds in New Zealand. Eradication of rats from one island led to a twenty fold increase in reptile numbers (Caughley et al, 1998). On Norfolk Island, black rats are the primary predators of the Norfolk Island green Parrot (*Cyanoramphius novazlandiae cookii)* and are considered to be a major threat to the endangered Norfolk Island Boobook Owl (*Ninox novaeseelandiae undulata),* (Olsen, 1996). Not only can they eat owl chicks, they may also occupy holes that might be important for nesting owls. They have even been known to take chicks from sitting females (Stevenson, 1997).

Invasive ability and rate of spread

Black rats can breed throughout the year. They usually have 3 litters per year but in a good season can produce 6 litters. The litter size varies from 5 to 10. The dispersive characteristics of black rats are similar to those for mice.

2.21 Brown rat, *Rattus norvegicus*

Current and potential distribution (Figures 21a and 21b, Appendix 2)

Brown rats are natives of south-eastern Asia, south of the Himalayas (Lever, 1985). They are now widely distributed throughout the world, mainly due to inadvertent introductions by people. Brown rats probably arrived in Australia soon after European settlement commenced in 1788. In Australia, brown rats are closely associated with human settlement where they are most common. They occur but are not as common as the black rat in the bush. Preferred wild habitat in Australia are along creeks in the wetter southern parts of the mainland and Tasmania.

Impact on native species

Little is known about the impact of brown rats on native wildlife, but they are not considered to have any near the same impact as black rats.

Invasive ability and rate of spread

Females can produce litters of up to 18 but usually 7 to 10. They have similar dispersive characteristics to that of the black rat and house mice.

2.22 Wolf snake *Lycodon capucinus*

Current and potential distribution (Figures 22a and 22b, Appendix 2)

The wolf snake occurs widely throughout Indonesia and neighbouring countries. It also now occurs on Christmas Island where it is assumed that it was accidentally introduced with imported goods (Cogger, 2000). However, Cogger suspects that they may have been introduced earlier either accidentally or deliberately, mainly because several adult and sub-adult specimens have been found over a short period. They are nocturnal, climbing snakes that are attracted to human dwellings. They are non-venomous and feed mainly on small reptiles such as geckoes and small mammals. Potentially, they could inhabit most of Christmas Island.

Impact on native species

Only a few specimens have been collected to date but if they become established, they could threaten Christmas Island's endemic reptiles and the islands only endemic mammal, the Christmas Island Shrew. An indication of the potential damage that the wolf snake could cause is shown by the impact that the Australian brown tree snake (*Boiga irregularis)* has had in Guam.

Invasive ability and rate of spread

Little is known of their invasive ability, but they could inhabit most of Christmas island as well as other Australian territories such as Cocos Keeling, should they also be introduced.

2.23 Grass skink *Lygosoma bowringii*

Current and potential distribution (Figures 23a and 23b, Appendix 2)

This lizard is widely distributed in south-east Asia. It is found only in disturbed habitats on Christmas Island where it appears to have been accidentally introduced recently (Cogger, 2000).

Impact on native species

Its likely impact on native wildlife is unknown, but if it continues to be restricted to disturbed habitat, it is likely to be minimal.

Invasive ability and rate of spread

This skink prefers disturbed habitat so its spread is likely to be limited to those areas. However, it could also occupy similar habitats on Cocos Keeling Island should it be accidentally introduced there.

2.24 House gecko, *Hemidactylus frenatus*

Current and potential distribution (Figures 24a and 24b, Appendix 2)

This gecko is widely distributed through the tropics including much of Indo-Malaysia, New Guinea and many Pacific Islands. It appears to have been accidentally introduced to northern Australia and to Christmas and Cocos Keeling Islands. On mainland Australia, it is restricted to several settlements around the northern coast from Darwin to Cairns (Cogger, 2000). It seems to prefer living in and around buildings. Similarly on Christmas Island, it is confined to buildings and vegetation in heavily disturbed areas.

Impact on native species

Given its preference for buildings and heavily disturbed sites, the house gecko is unlikely to be a serious threat to native wildlife.

Invasive ability and rate of spread

House geckos have a wide natural and naturalized distribution. They are closely associated with human occupied habitat and can quickly colonise such areas. For example, they have spread from Darwin along the human settlements towards Alice Springs (Ehrmann 1992). However, they also seem to disappear from long abandoned settlements. Their ability to colonise undisturbed or slightly disturbed habitats is debated (Ehrmann 1992).

2.25 Red-eared slider, *Trachemys scripta elegans*

Current and potential distribution (Figures 25a and 25b, Appendix 2)

The red-eared slider is probably one of the most commonly kept reptiles in the world. It is native to the Mississippi drainage of the USA (Ernst et al, 1994). However, they have established feral populations in many parts of the world, either by deliberate release or dumped or escaped pets. It is mainly aquatic, but lays its eggs or land and regularly moves between adjacent water bodies within their home range. They only reluctantly move further afield in response to pressures such drying of the water body. It prefers quiet fresh-water systems that have abundant aquatic vegetation and muddy bottoms. They will occupy farm dams, slow moving rivers, creeks and swamps. It can overwinter at temperatures below 10 C by becoming torpid in bottom mud, in hollow stumps and in disused burrows of other animals. They can survive even cool areas as long as there are good basking spots. Its current distribution in Australia is limited to a few creeks in the eastern and western suburbs of Sydney. Only a few specimens have been collected so far but they have the potential to establish in fresh water systems along most of the eastern and western coastal hinterland (John Cann, Sydney, pers. comm. 2000).

Impact on native species

Sliders are omnivorous with hatchlings and juveniles being more carnivorous feeding on small aquatic organisms such as snails and insect larvae. As adults they will eat insects, yabbies, shrimp, snails, worms, amphibians and small fish. Plant material includes algae, duckweed, and other aquatic plants.

There are no sound studies that quantify the damage that sliders cause. Currently, they are in relatively low density. It is likely that they occupy systems that are already degraded, but they have the potential to impact on native vegetation and wildlife. In aquariums, red-eared sliders out-compete native short-necked turtles for food but not native long-necked turtles.

Invasive ability and rate of spread

Sliders are prolific breeders. Males are sexually mature between 2-5 years and females between 5-7 years. In a good year they can lay two or even three clutches of between 5 to 20 eggs. Animals have been recorded up to 40 years in captivity, but probably live much less in the wild.

3.0. Threat category for introduced established reptiles and mammals

As stated in section 1.0, the information on the impact that pest animals have on the to conservation of biodiversity is generally poor. Consequently, it is difficult to make accurate assessments of the conservation threat posed by each pest species. In making the assessments concerning the risks posed by pest animals, it is recommended that a risk management process be adopted.

Ideally this should incorporate a comprehensive benefit/risk analysis. Risk analysis involves identifying potential benefits as well as the damage and other undesirable outcomes and the mechanisms that cause them and then estimating the probability that they will occur and their consequences. It may also include assessing the risk of taking no action about the potential hazard and the risk in following a particular course of action based on the potential hazard and the risk in following a particular course of action (See Bomford, 1991). However, a full risk/benefit analysis for each species is beyond the scope of this report.

In the absence of a full risk/benefit analysis, established mammals and reptiles have been categorised into the following categories (Table 2) based on an assessment of the available information and summarised in section 2:

- **Extreme threat (E).**
- **Serious threat (S).**
- **Moderate threat (M).**
- **Low or no threat (L).**

There are several ways by which pest animals may threaten the conservation status of Australian plants and animals (See Olsen 1998). The following are some key examples:

- Impact on endangered and threatened species through direct predation or competition.
- Damage to the habitat and other essential resources required by native plants and animals.
- Impact on other native species including communities.
- Impact on ecosystem processes.
- Impact on non-target wildlife due to the application of control strategies for the exotic species.

The impact of pests and hence their relative pest status varies considerably across their distribution. This largely reflects the usually patchy and fragmented distribution of the plant and animal species threatened by pest animals. In some areas the natural habitat is so degraded or the key native species are not present so that the pest has minimal or no impact. For example, feral and free-range domestic cats probably have little impact on the survival of threatened or endangered wildlife in most of urban and peri-urban Australia. A long-term study of bird species and numbers in a newly developing suburb in outer Canberra, showed that the number and density of native bird species increased several fold over a twenty year period despite the presence of wild and domestic cats (David Purchase, Canberra, pers. comm., 1986). David concluded that the increased diversity and density of native birds was due to improved habitat resulting from the establishment of native urban gardens.

The threat that each pest is to a particular native species, communities and ecosystem processes needs to be assessed on a case by case basis. Setting priorities at a regional/local scale will help managers make best use of available resources to address the adverse impacts of pest animals (see section 4.1).

Table 2. Suggested threat category for alien mammals and reptiles

Common name	**Scientific name**	**Comments**	**Threat Class. (E , S, M or L)**
European rabbit	*Oryctolagus cuniculus*	Nationally recognised key threatening process.	**E**
European red fox	*Vulpes vulpes*	Nationally recognised key threatening process.	**E**
Feral goat	*Capra hircus*	Nationally recognised key threatening process.	**E**
Domestic cat	*Felis catus*	Nationally recognised key threatening process. Well documented impact to wildlife on islands and threat to re-introductions of threatened natives. Elsewhere, impact is not well documented.	**E**
Black rat	*Rattus rattus*	Black rats are only an extreme threat on islands where they can severely threaten breeding sea birds.	**E**
Feral pig	*Sus scrofa*	Damage to wildlife is not well quantified. Are widely distributed, through extensive natural habitat. Likely to severely affect the long-term conservation of some native plants and animals in tropical and temperate wetlands.	**E**
Feral horse	*Equus caballus*	Contribute to overall grazing impact due to other exotics, stock and native grazers.	**S**
Feral donkey	*Equus asinus*	Similar impact to horses.	**S**
House mouse	*Mus domesticus*	While not a serious direct threat to wildlife, legal and illegal techniques used to control plagues can cause significant losses of wildlife, particularly native birds.	**S**
Water buffalo	*Bubalus bubalus*	In their current distribution, are a serious threat. Should they be allowed to reinvade their previous range, they would become an extreme threat.	**S**

Table 2. Suggested threat category for alien mammals and reptiles (continued)

Arabian camel	*Camelus dromedarius*	Similar but less impact than horses.	**M**
Brown rat	*Rattus norvegicus*	Mainly on islands and localised coastal wetlands.	**M**
European hare	*Lepus capensis*		**M**
Sambar	*Cervis unicolor*	Continuing to spread in the eastern highlands, often in fragile bush.	**M**
Feral dog/Dingo	*Canis familiaris*	Generally regarded as native but cause wildlife damage on small reserves close to settled areas.	**M**
Red-eared slider	*Trachemys scripta elegans*	Very limited distribution, but potential to spread widely.	**M**
Three-striped palm squirrel	*Funambulus pennanti*	Restricted distribution – likely damage unknown.	**M**
Wolf snake	*Lycodon capucinus*	Limited distribution, but if spread could have serious impact similar to brown tree snake on Guam.	**M**
Banteng	*Bos javanicus*		**L#**
Red deer	*Cervus elaphus*		**L**
Fallow deer	*Dama dama*		**L**
Hog deer	*Axis porcinus*		**L**
Chital deer	*Cervus axis*		**L**
Rusa	*Cervus timorensis*		**L**
House gecko,	*Hemidactylus frenatus*		**L**
	Lygosoma bowringii		**L**

For the species given a low risk **(L)**, most are in low density and there is little information to suggest that they are a significant threat to native plant and animal wildlife.

4.0. Species of greatest threat and potential for management

4.1 Setting priorities

Rarely are one or more vertebrate pests the only factors threatening the survival of endangered native species, communities or ecosystem processes. It is usually a suite of factors that need to be addressed in an integrated holistic approach (see Braysher, 1993; Olsen 1998). For example, while Priddel and Wheeler (1990) showed that foxes and feral cats were key threats to the survival of Malleefowl, native raptors also took a significant number of young birds. However, survival of the chicks also depended upon the availability of high nutrient seeds which in turn were influenced by land management practices and the impact of native and exotic grazers. Long-term survival of Malleefowl requires all these factors to be addressed.

Hence when determining where action should be directed, it is usually more effective for managers to focus on the native species, community or ecosystem process that is under threat and to determine the suite of pest and other factors that need to be addressed rather than attacking the issue from the pest animal perspective. That is the focus should be on the conservation outcome. Several agencies have developed mechanisms for determining the priority species/communities for management action. For example, Environment Australia recently held a workshop on *'Protecting the Natural Treasures of the Australian Alps'* (Peter Coyne, Environment Australia, pers. comm., 2000). The aim of the workshop was to determine priority features and actions required to address them. A model was developed which included factors such as the Threat Status of the feature, e. g. native animal; its Distribution; Rarity; Impacts on the feature; and the Management Potential or potential for recovery through management.

A similar approach for other regions could help determine where action, including managing the threat from vertebrate pests would best be directed. Ideally the system for setting priorities should meet the following criteria:

- Be user friendly
- Transparent
- Robust, that is not influenced by small errors
- Be repeatable
- Meet the requirements of major stakeholders in relevant jurisdictions
- Sufficiently rigorous to produce realistic rankings
- Able to use existing data
- Suitable for a wide range of features and threats

Developing and applying a suitable process across the various bio-regions of Australia would be a major task and well beyond the scope of this report.

As an initial step in the process, it is recommended that key areas be identified where Extreme Threat pest animals do not occur and where there are native species, communities or ecosystem processes are likely to be under significant threat from these pests. Many of these areas are likely to be significant offshore islands such as Kangaroo Island or the south-west corner of Western Australia. Contingency plans should be developed and implemented to discourage the spread of extreme threat pests to these areas.

4.2 Management techniques

There are a limited range of techniques for controlling the damage pest animals cause to the conservation of native wildlife (Olsen, 1998). They include:

- Killing including poisons, traps and shooting.
- Removal by commercial or other harvest.
- Biological control, e. g. myxomatosis and RCD for rabbits and development of virally-vectored, immuno-sterility agents.
- Exclusion by fences and other barriers.
- Manipulation of habitat to make it less suitable for the pest and/or more suitable for the species under threat.

Usually, effective management of damage requires the strategic application of a range of techniques. Where practicable, managers aim to adopt pest management strategies that give long-term effective and efficient reduction in damage. Most of the established pests that are an extreme threat to native wildlife are associated with and well adapted to disturbed conditions, usually due to humans (see section 5.0). Animals such as rabbits, and foxes do not favour intact natural habitat. Hence, strategies that maintain or facilitate the enhancement of natural habitat are likely to not only reduce pest animal density and impact, but are also likely to favour native wildlife. For example, revegetating excess tracks and other access corridors in dense forest is likely to reduce fox density (Catling and Burt, 1995). Collapse of rabbit warrens and encouragement of native grass cover effectively prevented the re-invasion of rabbits to an important suspended native swamp in Namadgi National Park, Australian Capital Territory (Author's personal experience). However, there is also a danger in this approach. There have been cases where eradication of rabbits from islands has resulted in unexpected and damaging changes. For example, on Bowen Island, Jervis Bay, removal of rabbits to provide more breeding space for shearwaters *(Puffinus spp.)* and little penguins *(Eudyptula minor)* resulted in the eruption of exotic kikuya grass. Chemical control of this grass was required to enable penguins to reach their nesting burrows (Author's personal experience).

In Western Australia, fox predation of western ring-tailed possums, (*Pseudocheirus occidentalis)* (P. de Torres, CALM, WA, pers. comm., 1993) was reduced by encouraging the closing of the forest canopy. This enabled the possums to move between trees without taking to the ground where they were vulnerable to foxes and cats.

4.3 Management options

There are three basic management options for managing the damage due to established pests; eradication, short or long-term management and no management. See Olsen, 1998 for a discussion. This report has been asked to briefly address two options, eradication and containment.

Eradication

Eradication is the permanent removal of every individual. It is rarely practicable on the mainland except locally, due to eventual re-invasion of pests from surrounding areas. For eradication to be practicable, three essential and usually three desirable criteria need to be met (Bomford and O'Brien, 1995; Appendix 3). Usually this is only possible for islands and where a permanent barrier can be erected and maintained such as at Heirisson Prong in Western Australia. In the latter case, the peninsula or prong was separated from the mainland with a fox and cat-proof fence. Foxes and cats were eradicated from the core area and are continually controlled in a buffer zone (Danielle Risbey, pers. comm., Murdoch University, Western Australia, 1998).

The potential for eradication needs to be determined on a case-by-case basis. Pests that have and could be eradicated on islands and in isolated areas include rabbits, feral goats, feral cats and foxes.

Containment.

Pests may be contained in one area and prevented from invading an area containing threatened or vulnerable wildlife by erecting and maintaining a suitable barrier. It may be:

- a fence as discussed above at Heirisson Prong;
- a water barrier;
- maintenance of a broad band of habitat that is unsuitable to the pest (for example, a band of dense forest); or
- through the continued application of an effective management technique such as starling control on the Nullabor Plain, Western Australia or ongoing aerial baiting over a broad scale as conducted for fox control in Western Australia.

In all cases of containment, ongoing maintenance and monitoring is essential. To determine whether containment is a suitable option the following factors need to be considered:

- the level of current and future resources available for pest management;
- the reduction required in the pest population to achieve the desired reduction in damage; and
- the availability and practicability of pest management techniques.

If the resources necessary to effectively maintain the program are not available for the foreseeable future, then the strategy may ultimately fail as the pest re-invades or builds up in density again once control is relaxed.

Management of pest animal damage should be assessed based on the National Feral Animal Control Strategy (See Olsen, 1998 and the BRS managing pest animals series). Each situation needs to be assessed separately and, where appropriate, a local management plan developed and implemented in conjunction with key stakeholders. Potentially, all the established pests are amenable to this approach in some areas. For example, water buffalo could be contained to their now relatively small distribution by the strategic application of containment

5.0. Summary of known failures – reasons for success or failure of introduction

There have been approximately 47 species of mammals introduced into Australia, Table 3 (Myers, 1986). They include 16 species of deer, gold-spotted mongoose, ferrets, rabbits, hares and foxes. Of these, 59% successfully established wild populations. Mary Bomford (BRS, pers. comm., 2000) recently assessed 13 factors for which there was sufficient available information, to determine the extent that they contribute to the chance that an introduced vertebrate species (birds and mammals) may successfully establish in Australia. She found that only seven factors were significantly important in determining the success of past introductions. These were:

1. The minimum number of individuals released (birds only assessed).
2. The minimum number of release events (birds only assessed).
3. The minimum number of release sites (birds only assessed).
4. The degree of climatic match of their overseas distribution to climate in Australia.
5. The size of their overseas distribution.
6. Whether they had established exotic populations overseas.
7. Their association with humans and adaptation to human modified habitats.

Of these, Dr Bomford found that four factors (4, 5, 6 and 7) were by far the most important factors in determining the likely success of an alien species establishing in Australia.

The other factors that she assessed but found were not significant were:

- Mean clutch or litter size.
- Whether the animals produced more than one clutch/litter per year.
- Whether they were migratory.
- Whether they showed flocking/herding behaviour.
- The age at which they reach sexual maturity.
- Whether they had a broad or specialised diet.

Looking at the factors:

Minimum number of individuals released (1)*, number of release events* (2) *and release sites* (3).

Overseas studies have shown that the above factors play an important role in the success or failure of introductions (Lever, 1985). This is not surprising based on ecological principles. Chance events such as fire, or accidents are more likely to affect the survival small populations than larger ones. Also small populations are likely to be subject to other factors such as population inviability due to inbreeding, failure to find a mate, the increased risk of predation or inability of only a few animals being able to maintain structures such as burrow systems. Survival of a vertebrate population is likely to be low for populations below 20 individuals (Bomford, BRS, pers. comm., 2000). However, this is not necessarily always the case. There are many examples where less than ten animals have resulted in successful establishment. For example, Himalayan Thar and stoats in New Zealand (Lever, 1985).

Similarly, increasing the number of release events and the number of release sites will increase the chance that a population will not die out by chance events.

Climate matching (4) *and wide overseas distribution* (5)

Mary Bomford found that the degree of climate matching was the one of the strongest tools available for attempting to quantitatively predicting the likelihood of an exotic species establishing in Australia. Those species that have a wide overseas distribution and hence also are likely to find similar climate to their overseas. However, other factors are also important. For example, hares and feral pigs do not occupy all the habitat that climate matching based on their overseas distribution might predict. They are limited by other factors such as the availability of cover and access to free water in the case of feral pigs and probably due to dingo predation in he case of hares.

Established pest elsewhere (6)

88% of mammals that have successfully established in Australia have a history of successfully establishing exotic populations overseas (Mary Bomford, BRS, pers. comm., 2000). In contrast only 40% of mammals that failed to establish in Australia had a history of overseas establishment. However, care needs to be taken with this factor. Many species had not been introduced to other countries and so have not had the opportunity to demonstrate their invasive potential.

Adaptation to human modified habitats (7)

100% of the mammals that have successfully established in Australia can live in human modified habitats such as urban or agricultural areas. The same is true for the lizards that have successfully established in Australia. For example, the Asian House Gecko occurs no where else but in close association with humans (Cogger, 2000). Populations have died out when settlements have been abandoned.

Other factors

Myers (1986) concluded that of the 16 deer species that were introduced to Australia, only the less specialised species survived. Highly specialised species such as the primitive mouse deer (*Tragulus meminna)* which lives singly or in pairs in dense bush in the Himalayan forests of India and the small musk deer (*Moschus moschiferus)* from dense damp forests of Asia failed to establish. In contrast, the widely distributed fallow deer (*Dama dama)* that is well adapted to agricultural land was successful.

However, the failure for some species to establish is difficult to assess, mainly because the information concerning their introduction and release is not known. The well recorded attempts by Thomas Austin to introduce the European hare to his property near Geelong illustrates the problem (Rolls, 1969). Austin landed two pairs of hares at Melbourne in 1859, but unfortunately the two does died shortly afterwards. Of another pair sent in 1862, the buck died on the voyage out. There were similar failures of other species that failed to survive the rigours of the sea voyage or that were in such poor condition when they arrived that they had little chance of surviving. To overcome these problems, in 1863, Austin went to England and returned with 11 hares. They were put into individual pens on the ship and personally cared for by Austin. Only two were lost on the voyage. The others were transported to Geelong in covered hutches so that they would not be frightened by sightseers and released into a 4 acre dog-proof enclosure to acclimatise before eventually being gradually released to the wild. There is little information on the conditions under which other species were collected, transported and released so it is not possible to determine the extent to which their treatment and subsequent condition contributed to the success or failure of their establishment.

Table 3 Success of mammals introduced to Australia (after Myers, 1986)

Common name	**Scientific name**	**Result**	**Origin**
Eastern grey squirrel	*Sciuris carolinensis*	Died out	North America
Three-striped palm squirrel	*Funambulus pennanti*	Local population - Perth	India
Black rat	Rattus rattus	Widespread	S.E. Asia
Brown rat	Rattus norvegicus	Widespread	Asian steppes
House mouse	Mus domesticus	Widespread	Central Asia
Ferret	*Mustela putorius furo*	Died out	Mediterranean
Gold-spotted mongoose	*Herpestes javanicus aruopunctatus*	Died out	India, Java, Sumatra
Feral dog/Dingo	Canis familiaris	Widespread	Asia
European red fox	*Vulpes vulpes*	Widespread	Europe
Domestic cat	*Felis catus*	Widespread	Europe
European rabbit	*Oryctolagus cuniculus*	Widespread	Western Mediterranean
European hare	*Lepus capensis*	Widespread	Europe
Feral horse	*Equus caballus*	Widespread	Europe
Feral donkey	*Equus asinus*	Widespread	North Africa
Zebra	*Equus spp.*	Died out	Africa
Feral pig	*Sus scrofa*	Widespread	Eurasia
Arabian camel	*Camelus dromedarius*	Widespread	Arabia
Llama	*Lama guanaco lama*	Died out	High Andes
Alpaca	*Lama guanaco pacos*	Died out	High Andes
Vicuna	*Lama vicugna*	Died out	High Andes
Indian spotted mouse deer	*Tragulus meminna*	Died out	India
Musk deer	*Moschus moschiferus*	Died out	Central and East Asia
Fallow deer	*Dama dama*	Local herds	Mediterranean
Hog deer	*Axis porcinus*	Gippsland	India/Asia
Chital deer	*Cervus axis*	Local herds	India
Sambar	*Cervis unicolor*	Southern high country	India and Sumatra
Rusa	*Cervus timorensis*	Local herds	Borneo
Barasingha	*Cervus duvauceli*	Died out	India
Philippine sambar	*Cervus mariannus*	Died out ??	Philippines
Sika	*Cervus nippon*	Died out	Japan
Red deer	*Cervus elaphus*	Local herds	Europe
Wapiti	*Cervus elaphus canadensis*	Died out	North America

Table 2 Success of mammals introduced to Australia (continued)

Chinese water deer	*Hydropotes inermis*	Died out	China
Roe deer	*Capreolus capreolus*	Died out	Europe
Eland	*Taurotragus oryx*	Died out	Central Africa
Feral sheep	*Ovis ovis*	Widespread	Mediterranean
Feral goat	*Capra hircus*	Widespread	Mediterranean
Blackbuck	*Antilope cervicapra*	Died out ??	India
Water buffalo	*Bubalus bubalus*	Tropics	Asia
African buffalo	*Syncerus caffer*	Died out ??	Africa
Banteng	*Bos javanicus*	Local herds (Coburg Peninsula)	Java
Feral cattle	*Bos taurus*	Widespread	Europe
Zebu	*Bos inducus*	Local populations	India

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Appendix 1. Terms of reference

1. Summarise and review information on non-native mammals and reptiles covering:
- Their impact on native species, particularly nationally endangered or vulnerable species;
- Their effect on ecosystem functioning;
- Current and potential distribution; and
- Their invasive ability and rate of spread.
2. Using this information categorise them into broad band (extreme, serious, moderate and low) categories based upon their threat to the environment.
3. Identify those species that are considered of greatest threat, but may be amenable to containment or eradication and test the categorisation and list through relevant peer group review.
4. Provide a summary of known failures of introductions of non-native mammals and reptiles to establish long-term naturalised populations.
5. Provide a report covering the above scope items.

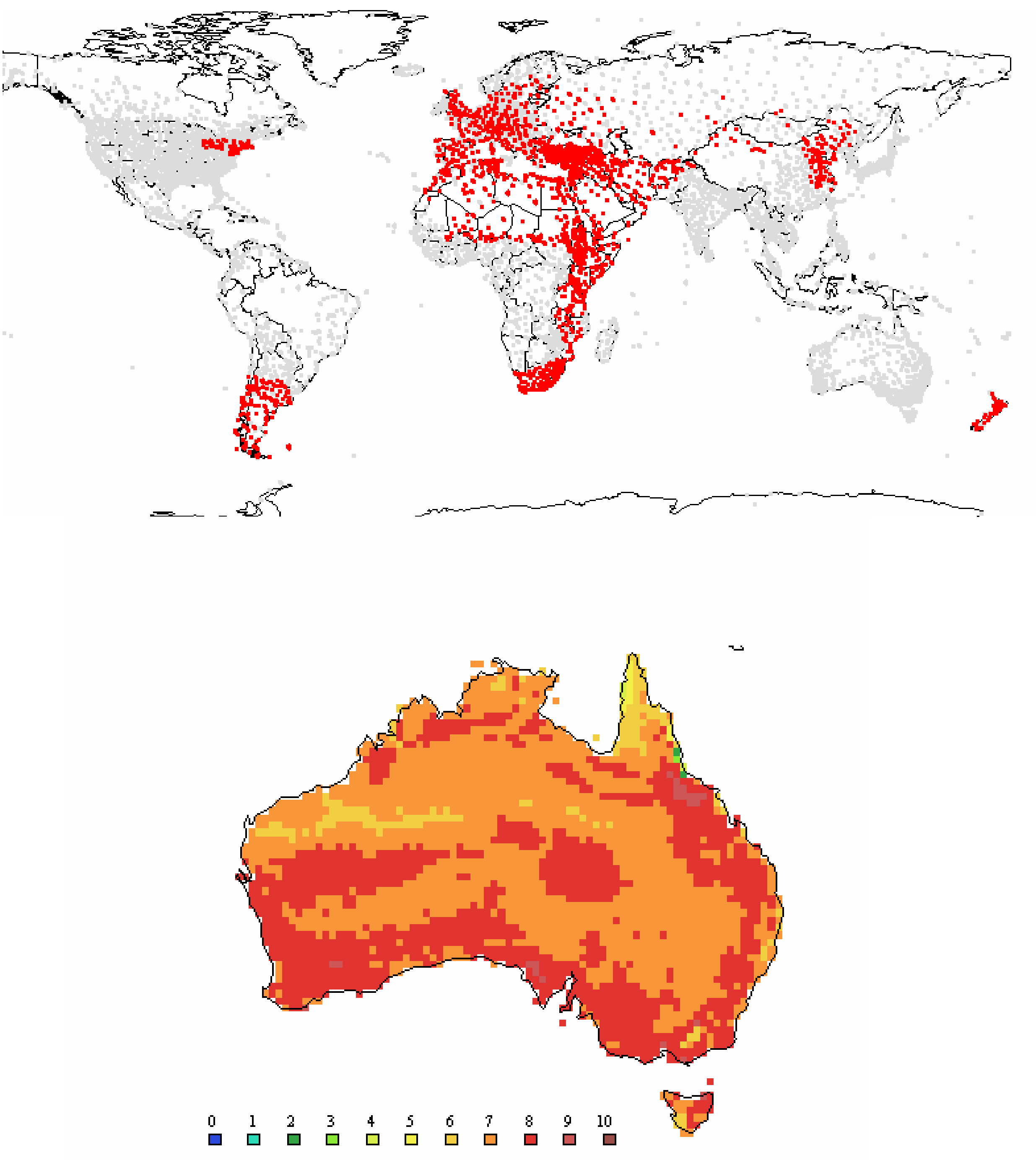
Appendix 2. Maps of climatically suitable habitat

The following table and figures were prepared by Dr Mary Bomford (Bureau of Rural Sciences, Canberra) and present CLIMATE matches between species' overseas ranges and Australia. The matches do not include the current Australian range. Including the current Australian range when conducting the matches could give improved predictions on where a species could spread to in Australia.

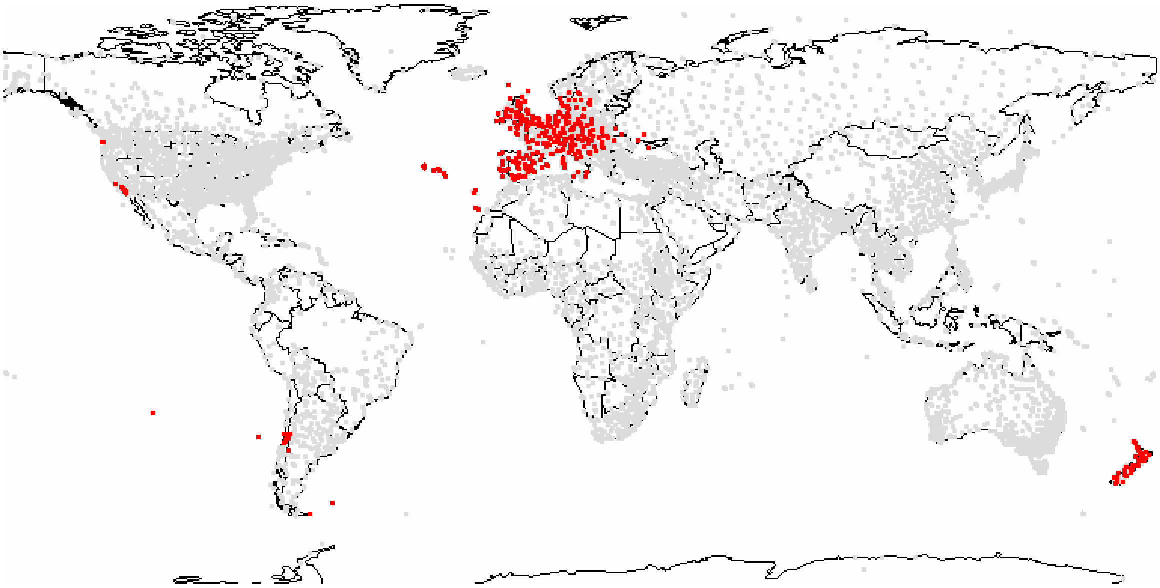
On the maps of Australia, level 9 (brown) represents the areas with the closest match to the climate in the species' overseas range, and level 0 (blue) represents the poorest match.

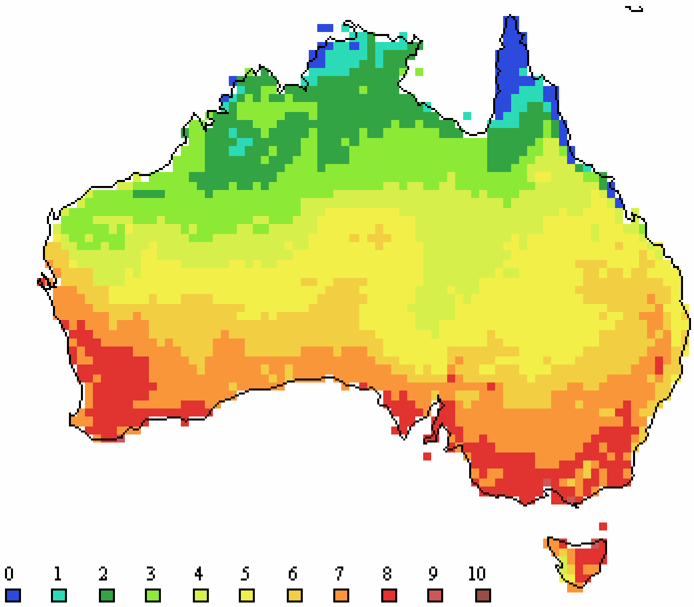
CLIMATE matches – Euclidian cumulative values. The PC version of CLIMATE never gives Level 10 matches when the Euclidian match algorithm is used so level Σ 9 is the highest (best) possible match for a species to Australia.

Species	Σ1	Σ 2	Σ 3	Σ 4	Σ 5	Σ 6	Σ 7	Σ 8	Σ 9	10
Brown hare, *Lepus capensis*	2785	2785	2783	2782	2779	2769	2637	1117	22	0
Rabbit, *Oryctolagus cuniculus*	2726	2668	2402	2037	1646	1067	659	241	4	0
European red fox, *Vulpes vulpes*	2785	2785	2785	2785	2784	2768	2274	520	3	0
Domestic cat, *Felis catus*	2785	2785	2785	2785	2784	2784	2764	1912	69	0
Feral horse, *Equus caballus*	2781	2771	2754	2706	2437	1742	898	199	2	0
Feral donkey, *Equus asinus*	2781	2771	2754	2706	2437	1742	898	199	2	0
Feral pig, *Sus scrofa*	2785	2785	2785	2785	2783	2777	2746	1318	19	0
Camel, *Camelus dromedarius* plus *C. bactrianus*	2782	2778	2773	2761	2756	2500	1616	214	0	0
Banteng, *Bos javanicus*	2509	2209	1866	1413	1060	802	415	65	0	0
Feral Asian water buffalo, *Bubalus bubalis*	2768	2591	2176	1699	1264	958	689	127	0	0
Indian Palm Squirrel, *Funambulus pennanti*	2710	2671	2611	2515	2163	1533	630	4	0	0
Axis, chital or spotted deer, *Cervus axis*	2783	2778	2770	2764	2748	2460	1657	328	0	0
Fallow deer, *Dama* (*Cervus*) *dama*	2724	2638	2543	2139	1636	1068	649	238	2	0
Red deer, *Cervus elaphus*	2783	2766	2529	2223	1978	1661	848	253	5	0
Hog deer, *Cervus porcinus*	2762	2418	1958	1437	1078	813	457	92	0	0
Timor or Rusa deer, *Cervus timorensis*	2579	1893	1452	925	480	255	109	25	0	0
Sambar, *Cervus unicolor*	2784	2784	2611	2115	1555	1132	784	169	1	0
Feral goat, *Capra hircus*	2785	2785	2785	2781	2745	2654	2023	373	3	0
House mouse, *Mus domesticus*	2785	2785	2785	2785	2785	2784	2774	2021	76	0
Black rat, *Rattus rattus*	2785	2785	2785	2785	2783	2783	2769	2004	69	0
Brown rat, *Rattus norvegicus*	2785	2785	2785	2785	2782	2779	2757	1575	61	0
Wolf snake *Lycodon capucinus*	2545	2254	1912	1448	1099	845	616	231	0	0
Grass-skink *Lygosoma bowringii*	2783	2765	2564	2199	1606	1208	775	236	0	0
House gecko, *Hemidactylus frenatus*	2783	2778	2768	2679	1856	1095	698	335	0	0
Red-eared slider, *Trachemys scripta*	2780	2776	2774	2762	2562	2110	1504	387	0	0

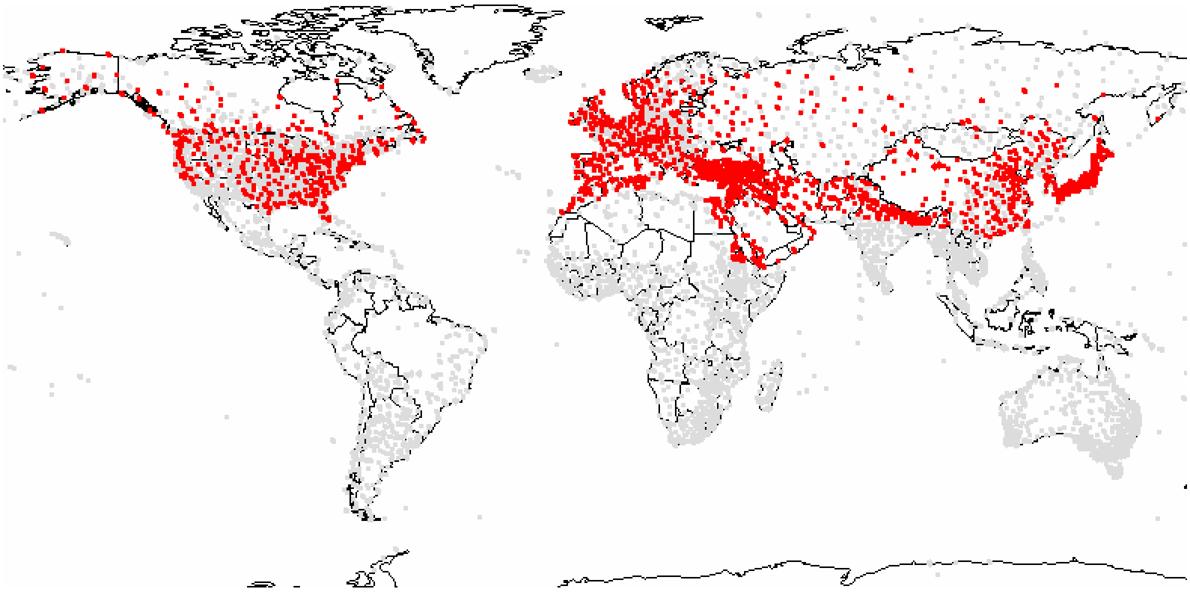


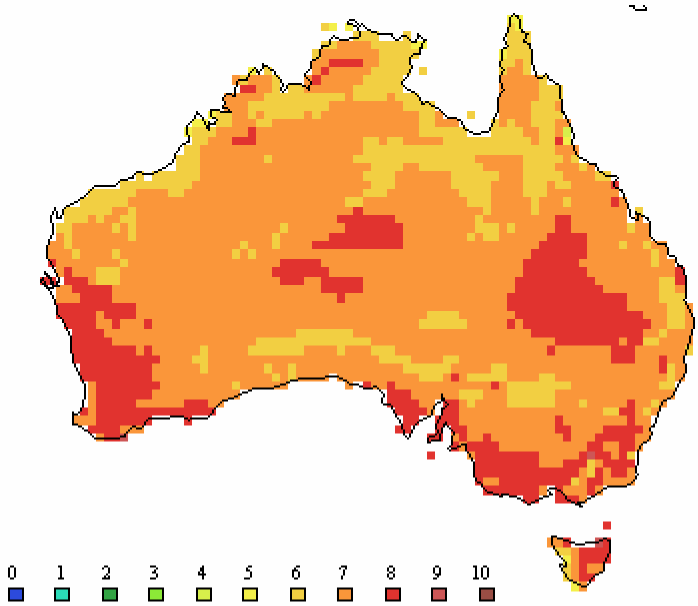
Figures 1a and 1b. Brown hare *Lepus capensis*



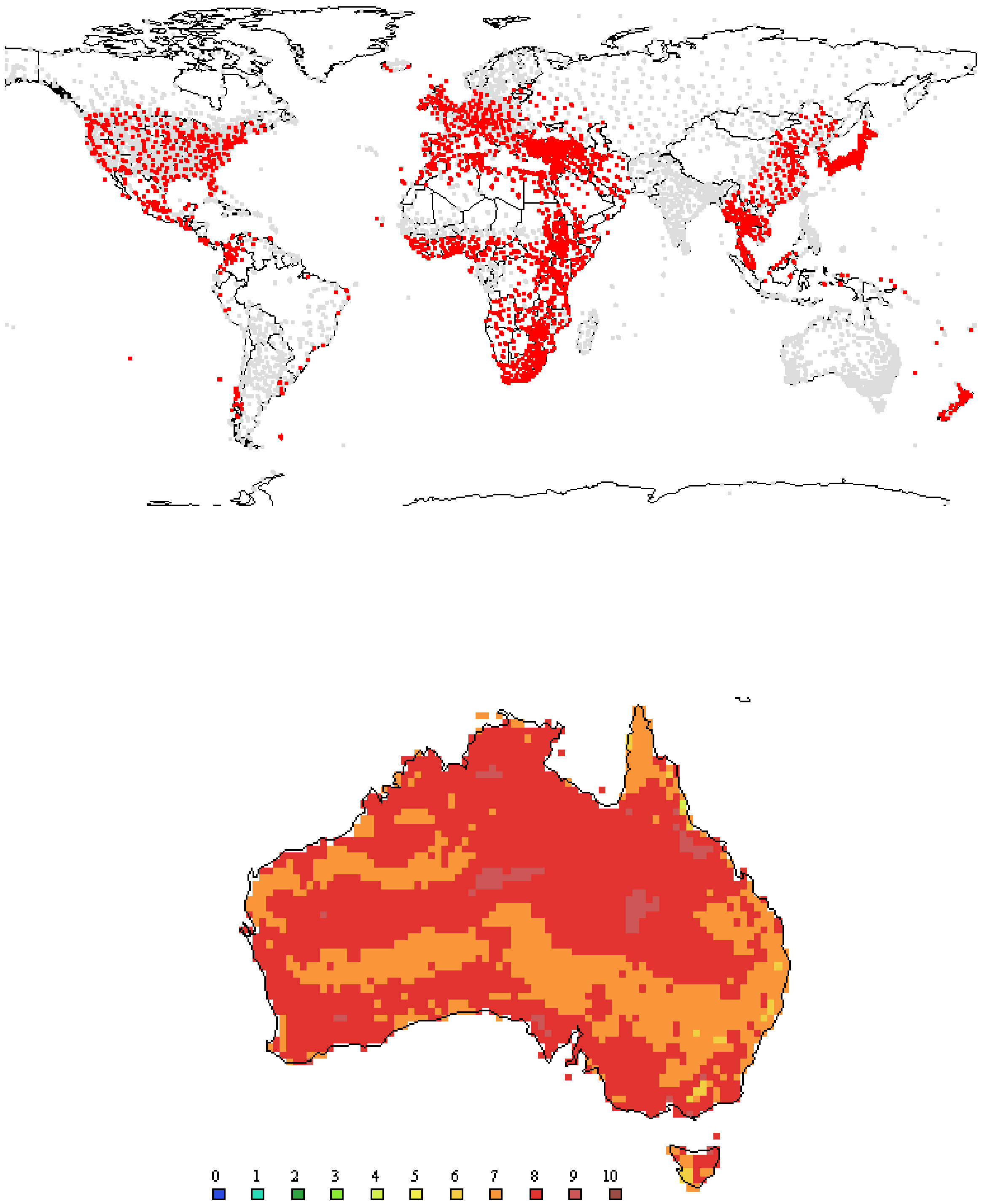


Figures 2a and 2b Rabbit *Oryctolagus cuniculus*

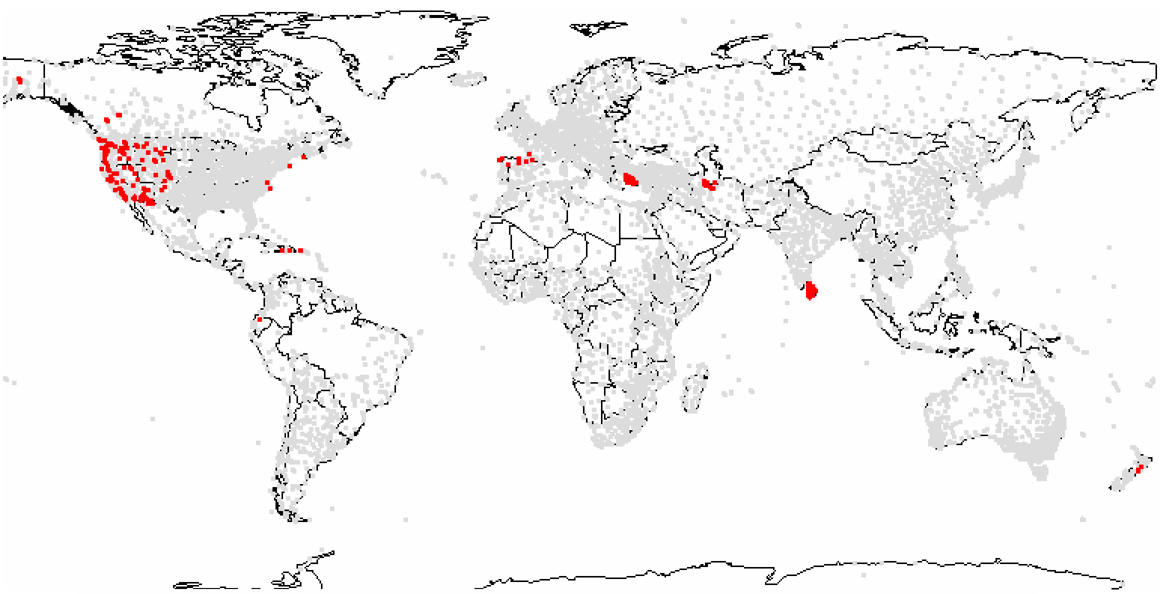


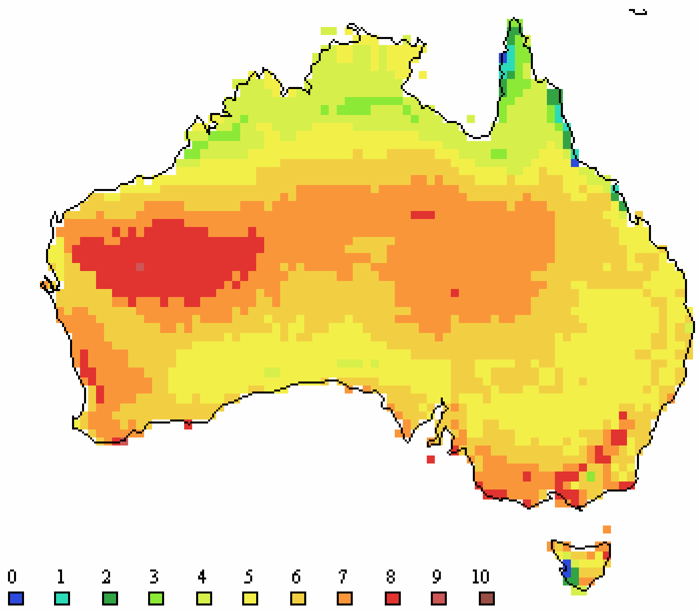


Figures 3a and 3b European red fox *Vulpes vulpes*

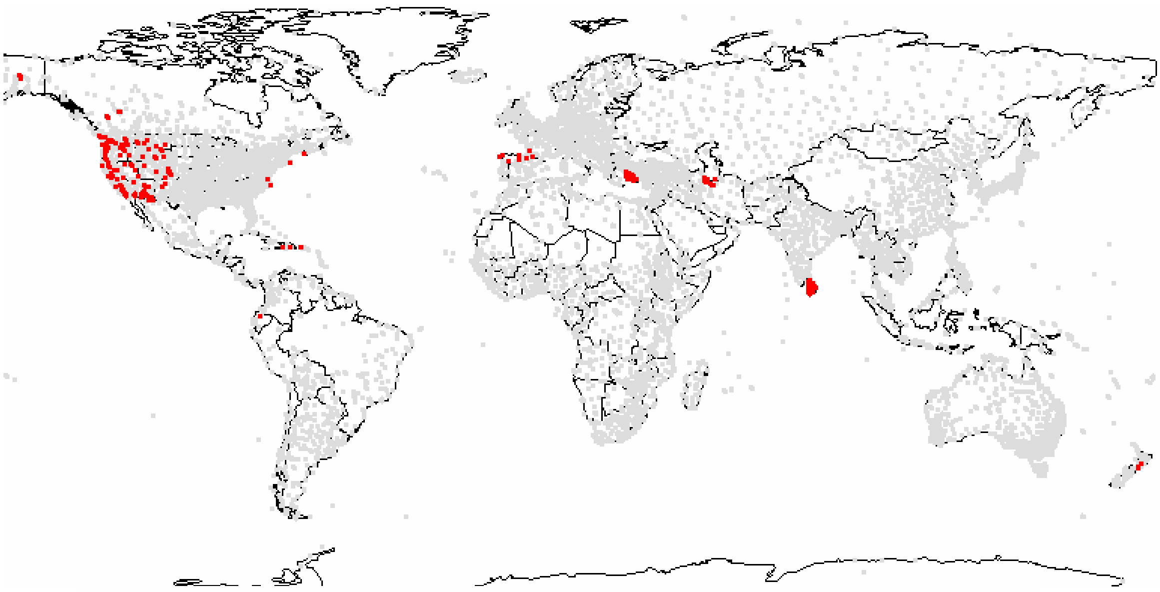


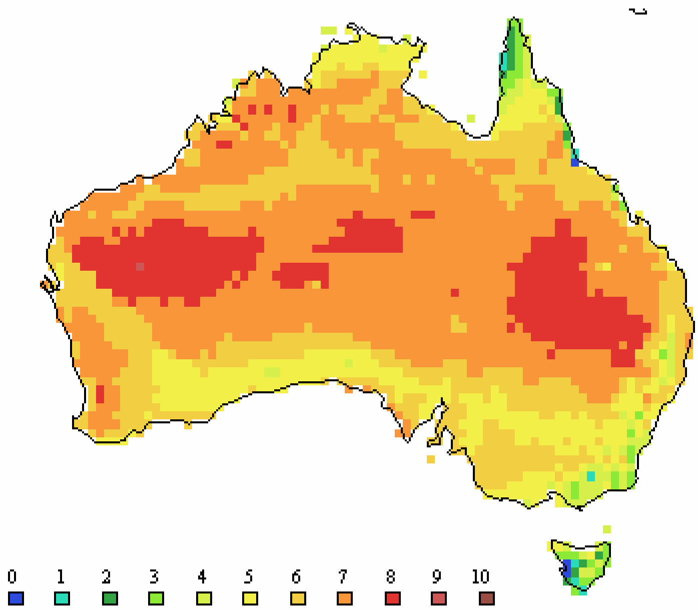
Figures 4a and 4b Domestic cat *Felis catus*



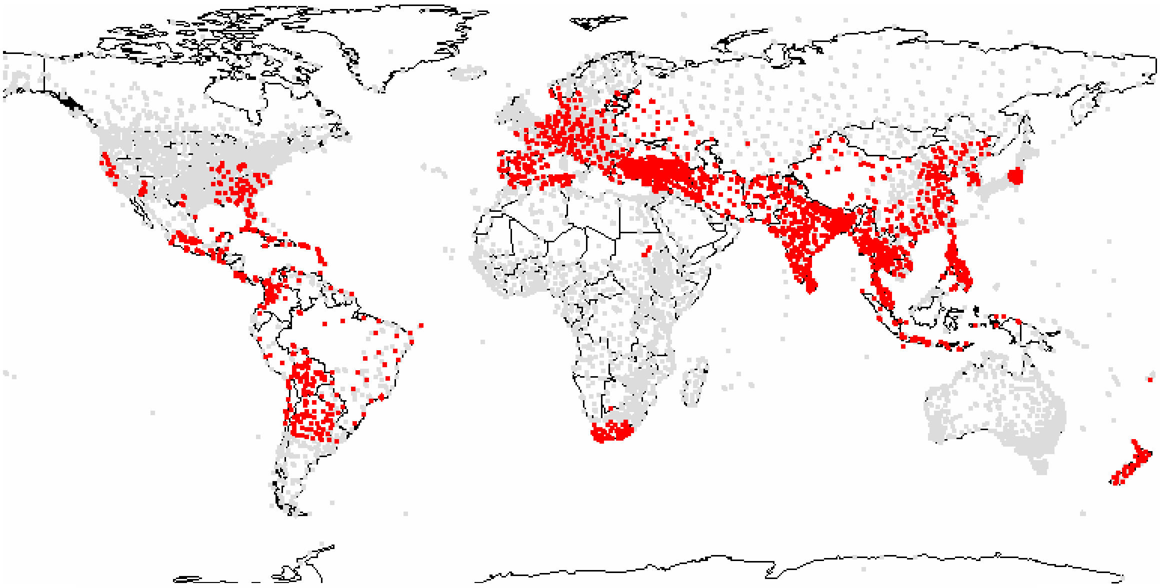


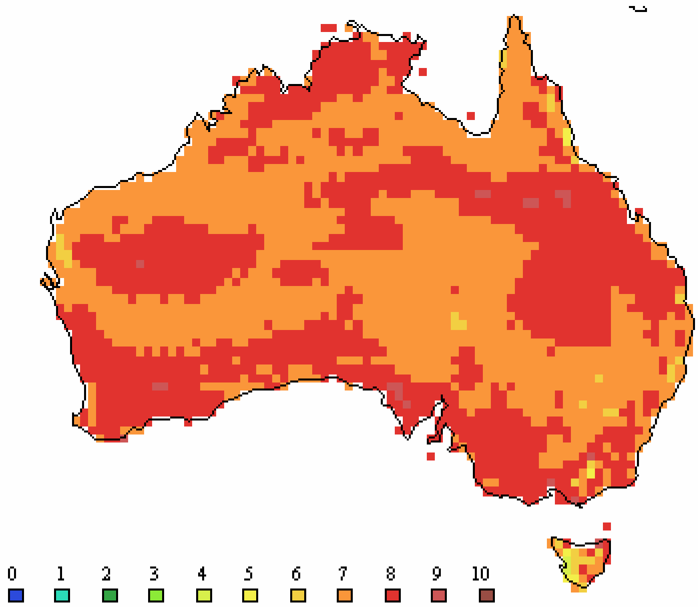
Figures 5a and 5b Feral horse *Equus caballus*



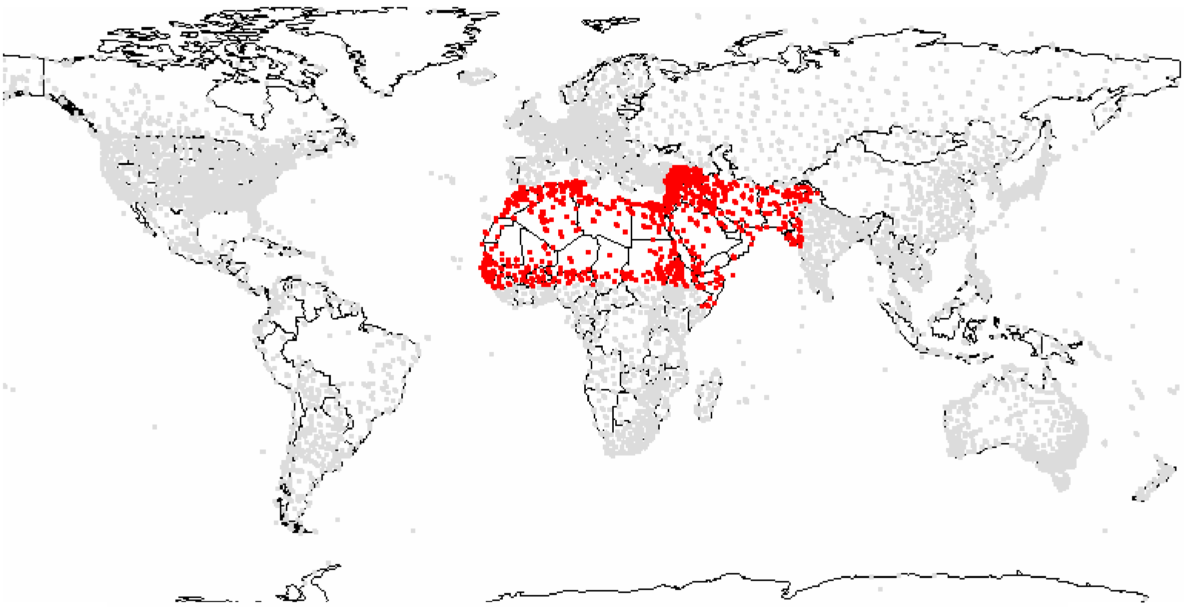


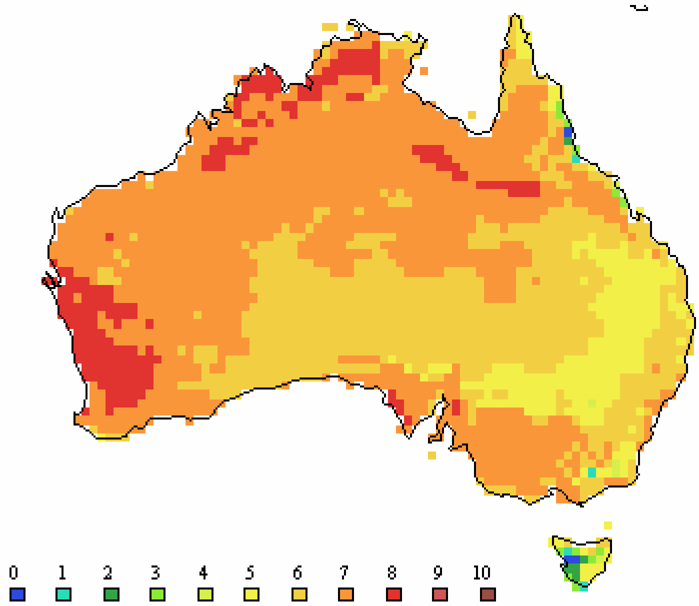
Figures 6a and 6b Feral donkey *Equus asinus*



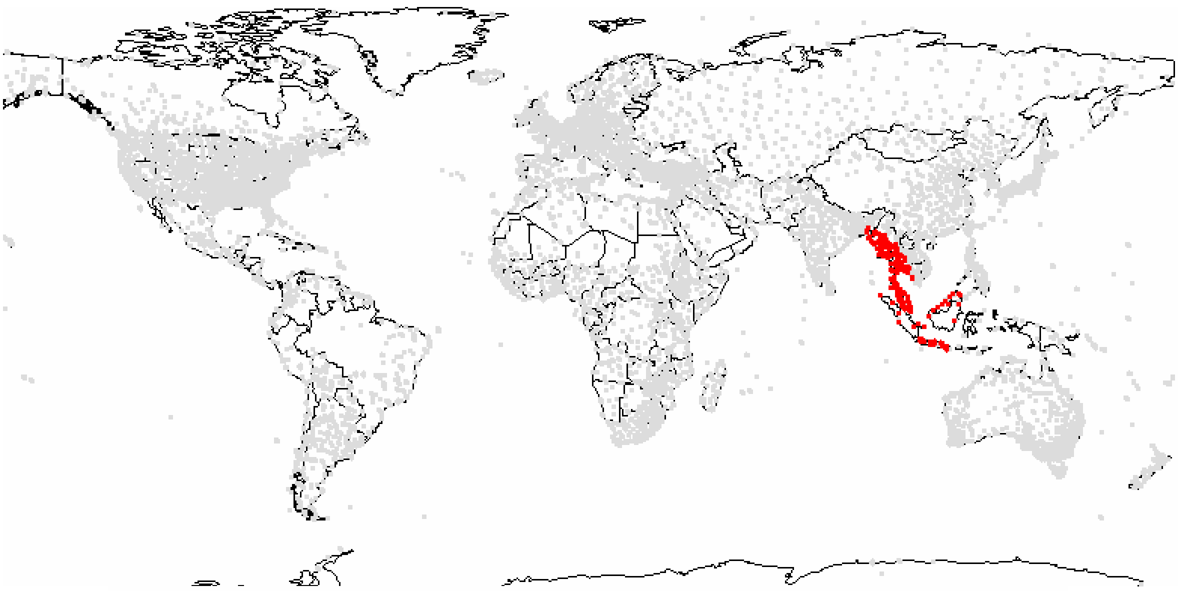


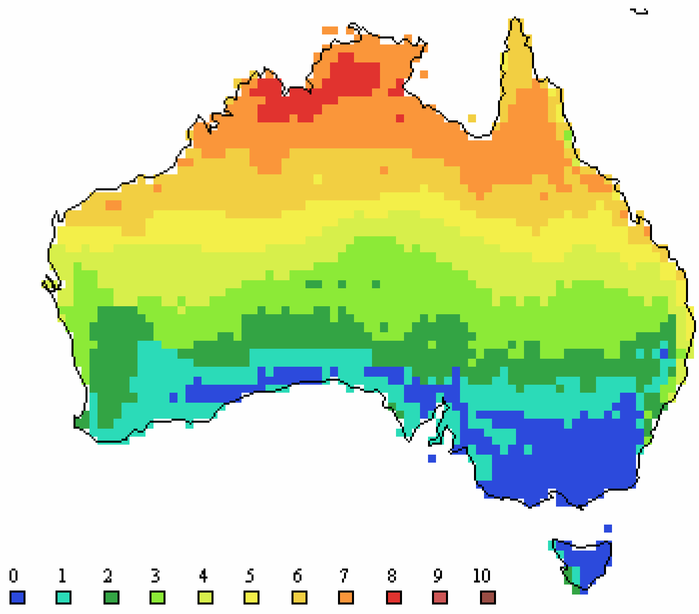
Figures 7a and 7b Feral pig Sus scrofa



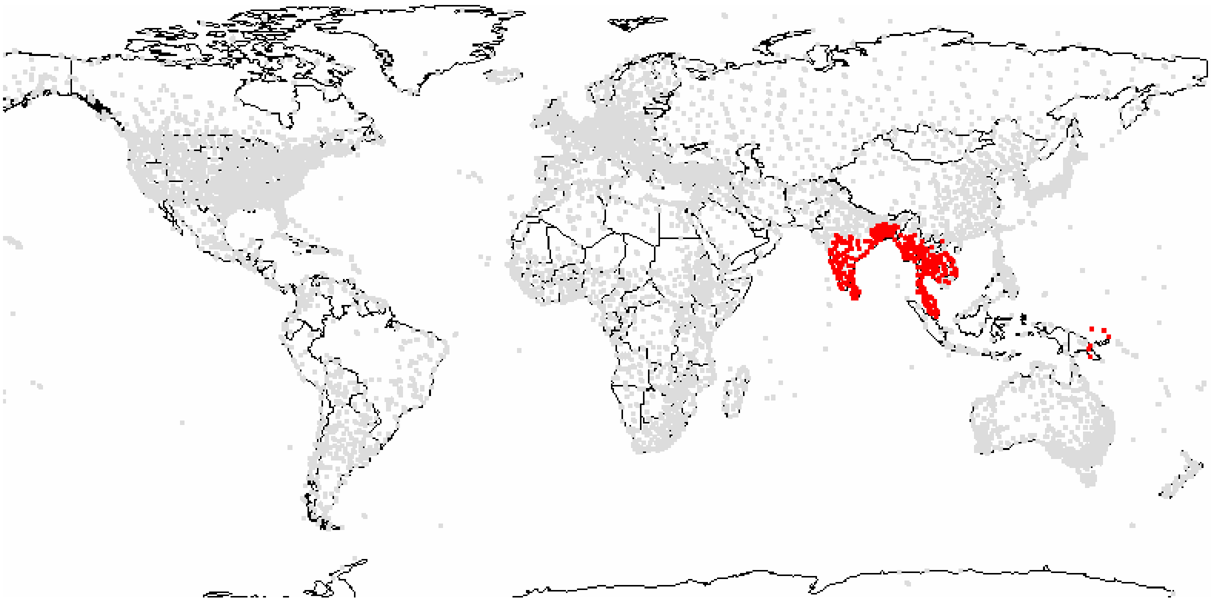


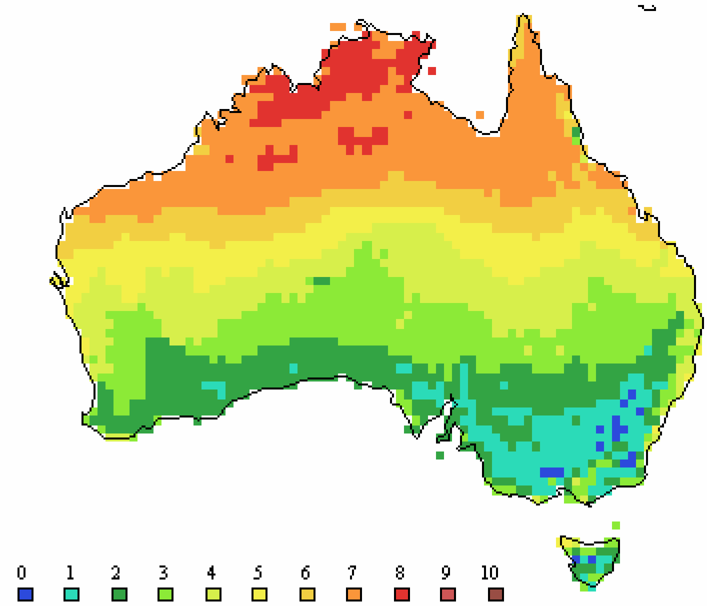
Figures 8a and 8b Camel, *Camelus dromedarius* plus *C. bactrianus*. According to Long (2003) *Camelus dromedarius* and *C. bactrianus* are both the same species — they can interbreed and have fertile offspring. There are no longer any wild populations of *C. dromedarius* other than in Australia. The input map for this species is the current range of *C. bactrianus* plus the former range of *Camelus dromedarius* as presented by Long (2003).



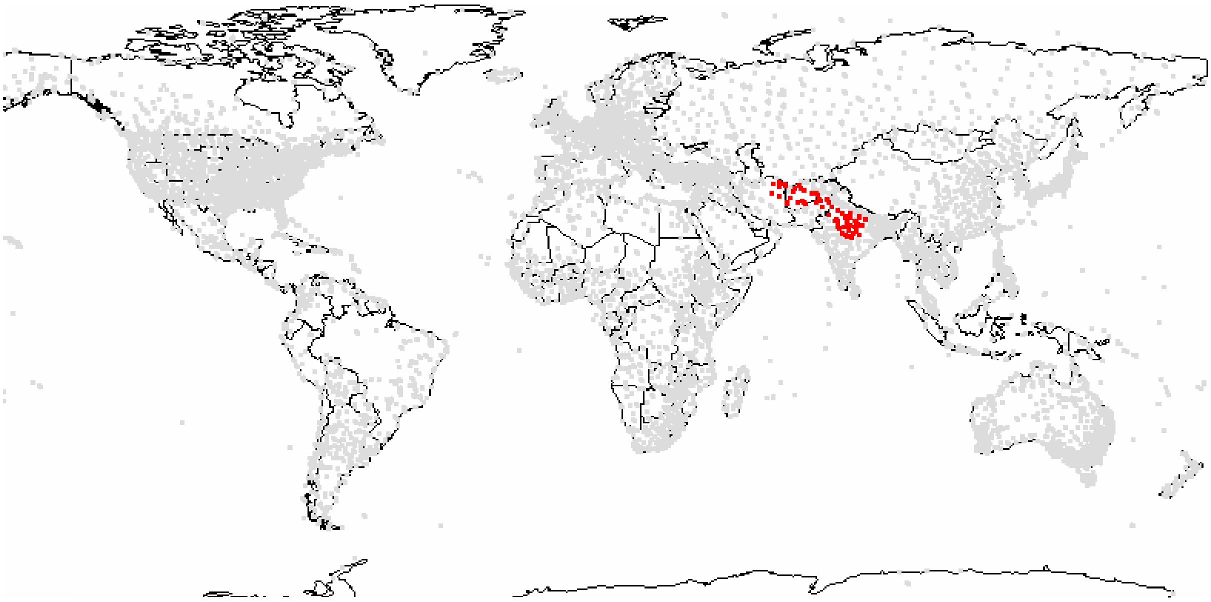


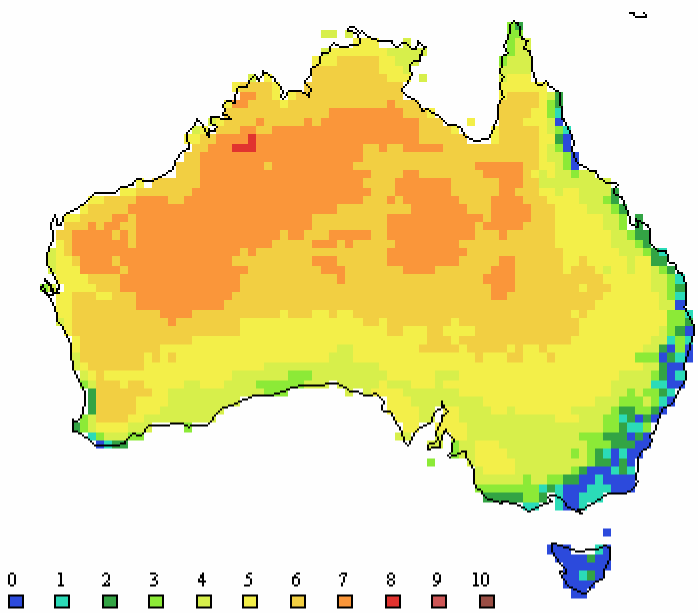
Figures 9a and 9b Banteng *Bos javanicus*



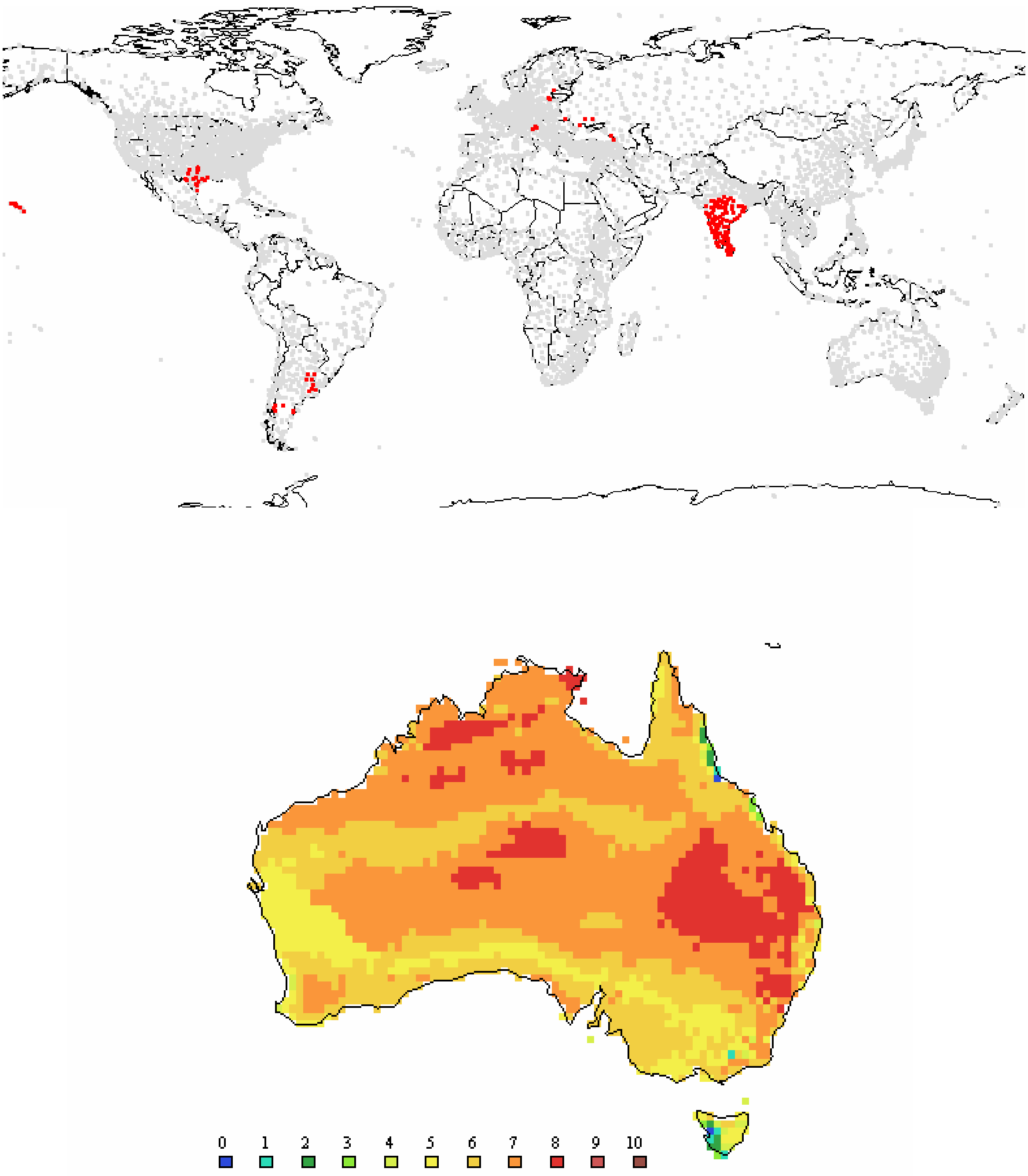


Figures 10a and 10b Feral Asian water buffalo *Bubalus bubalis*

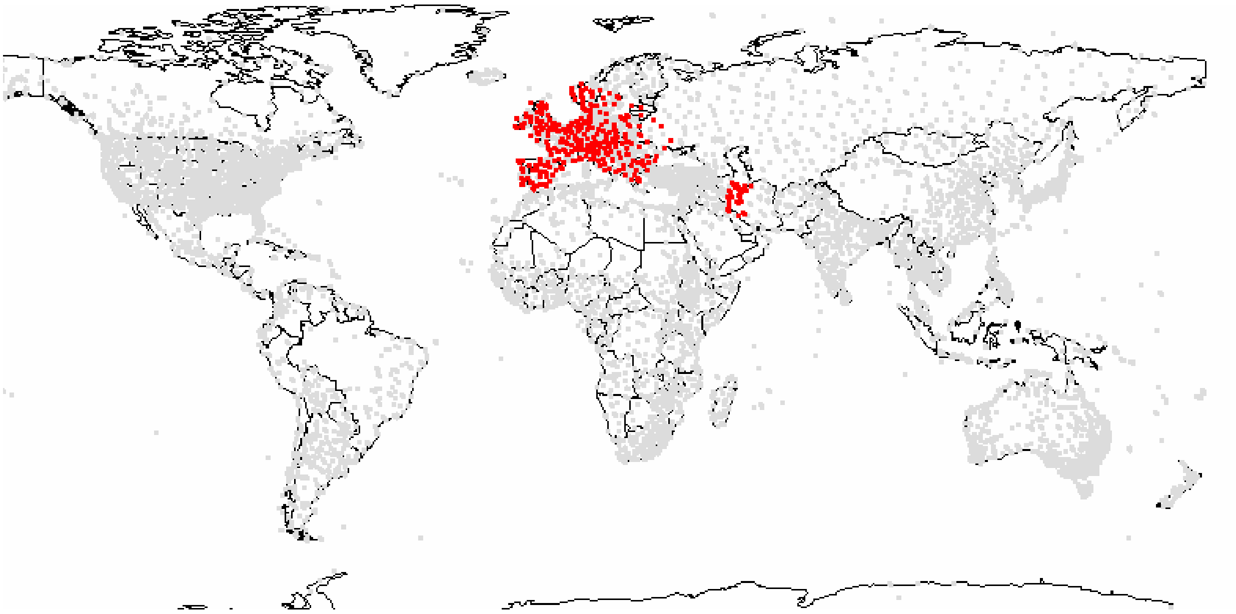


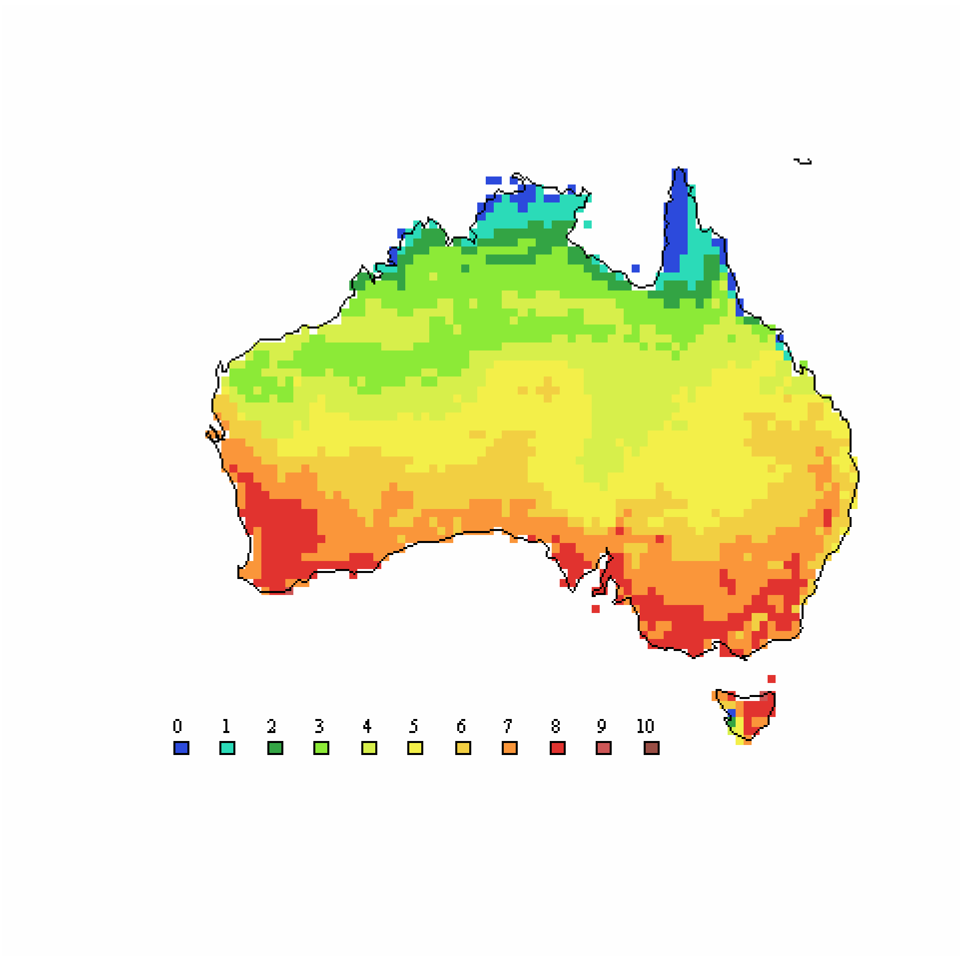


Figures 11a and 11b Indian palm squirrel *Funambulus pennanti*

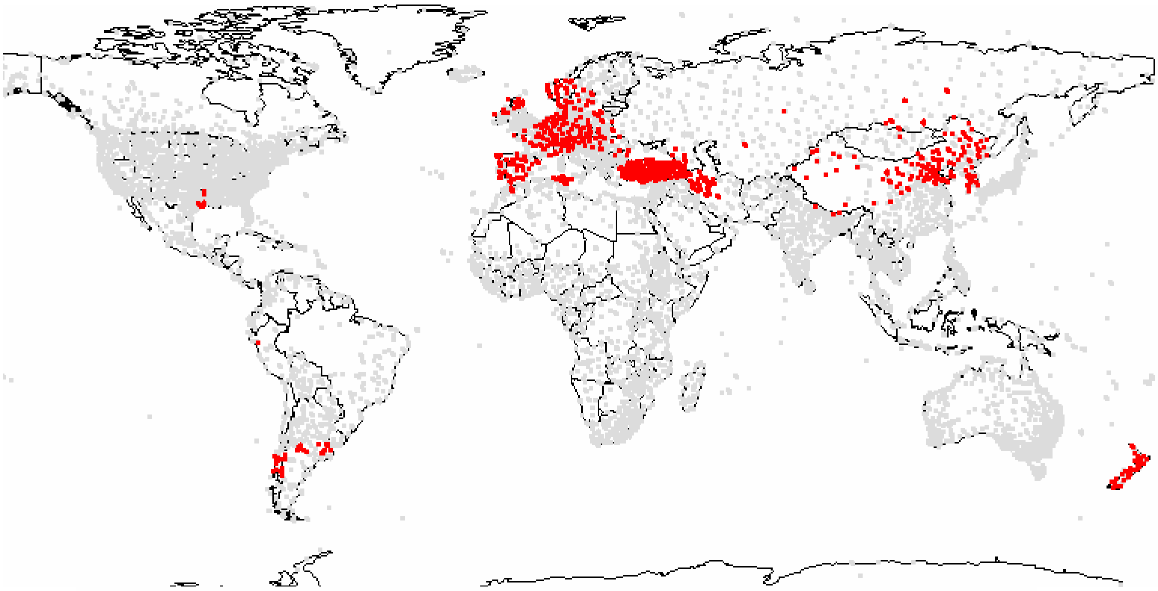


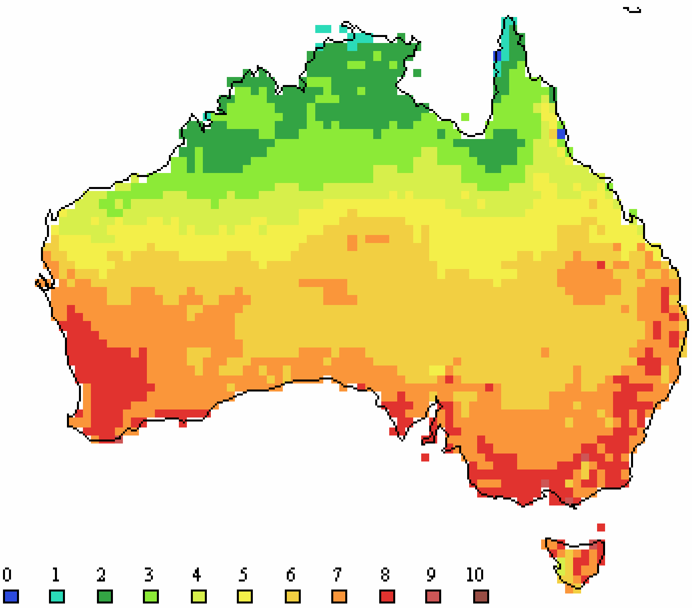
Figures 12a and 12b Axis, chital or spotted deer, *Cervus axis*



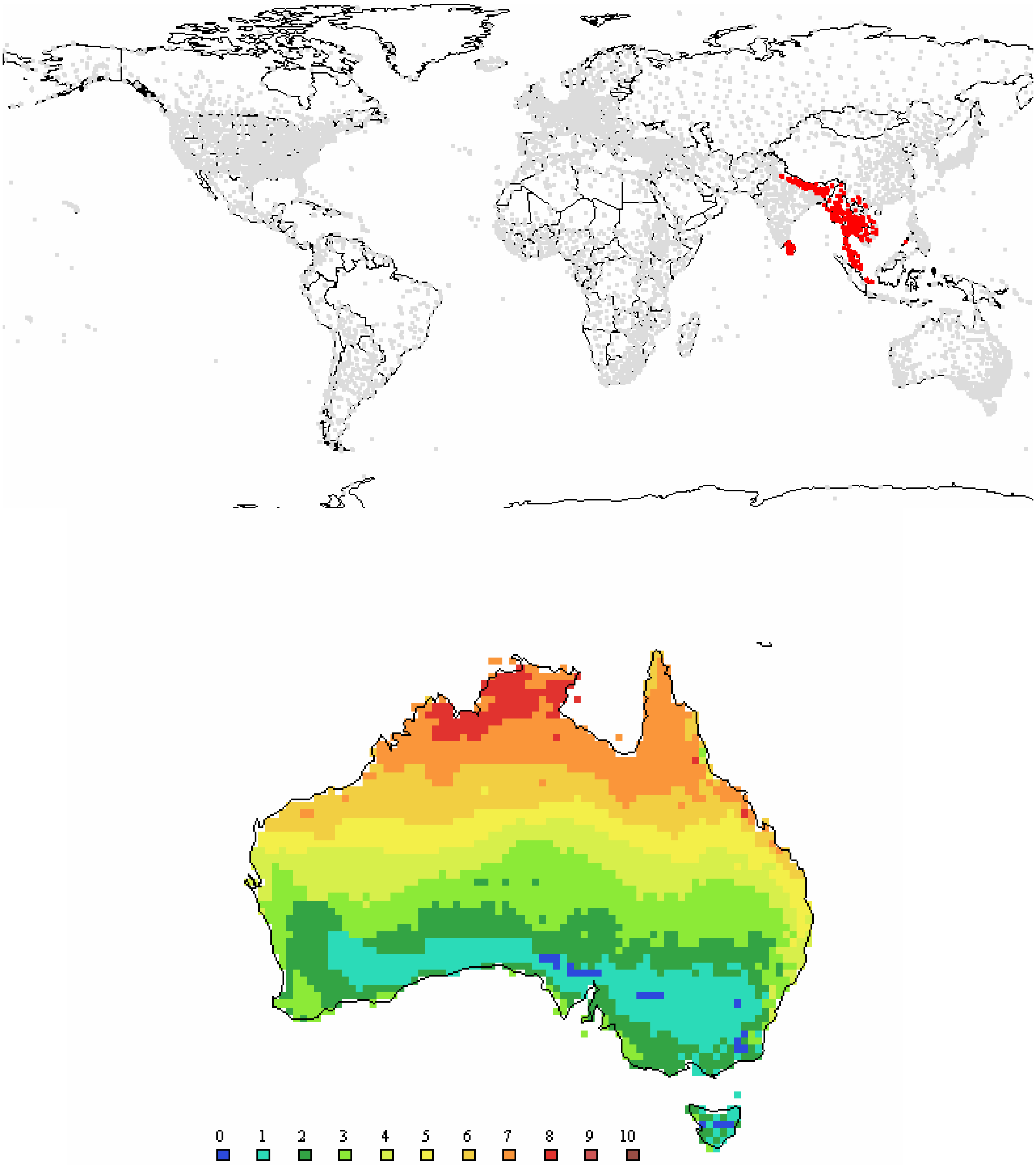


Figures 13a and 13b Fallow deer *Dama (Cervus) dama*

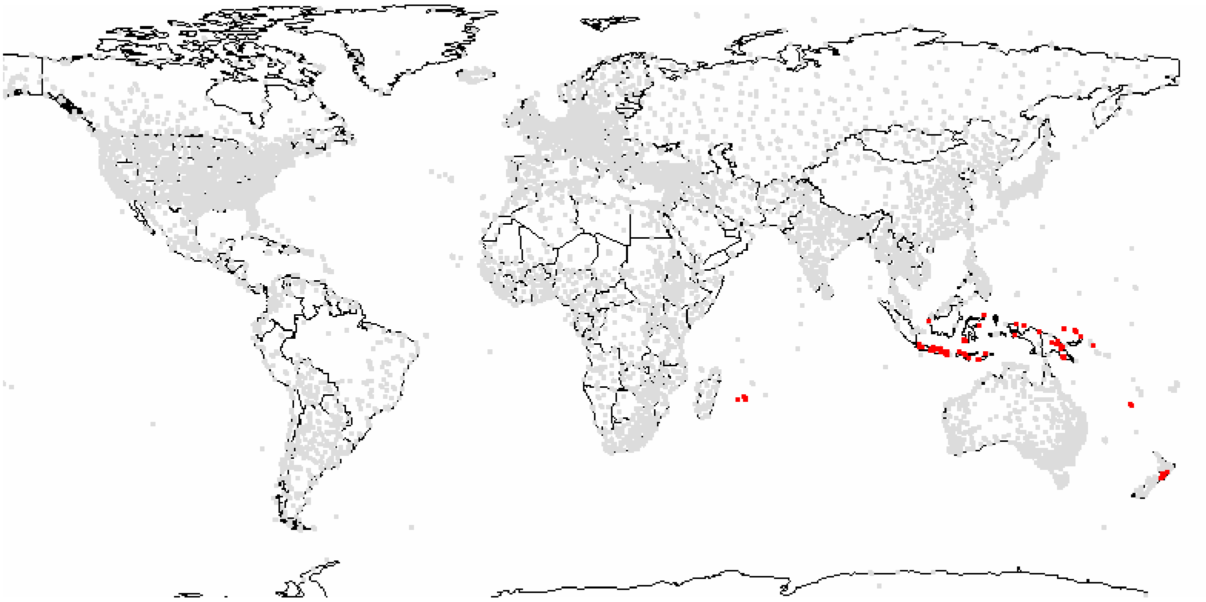


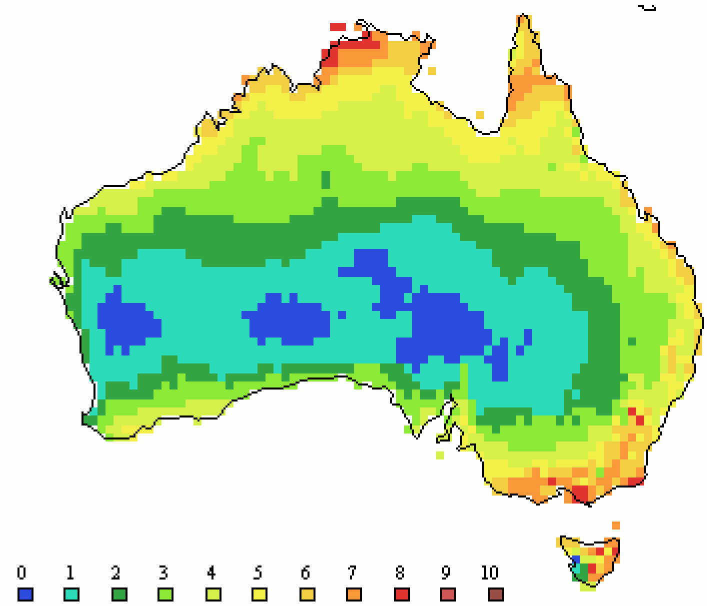


Figures 14a and 14b Red deer *Cervus elaphus*

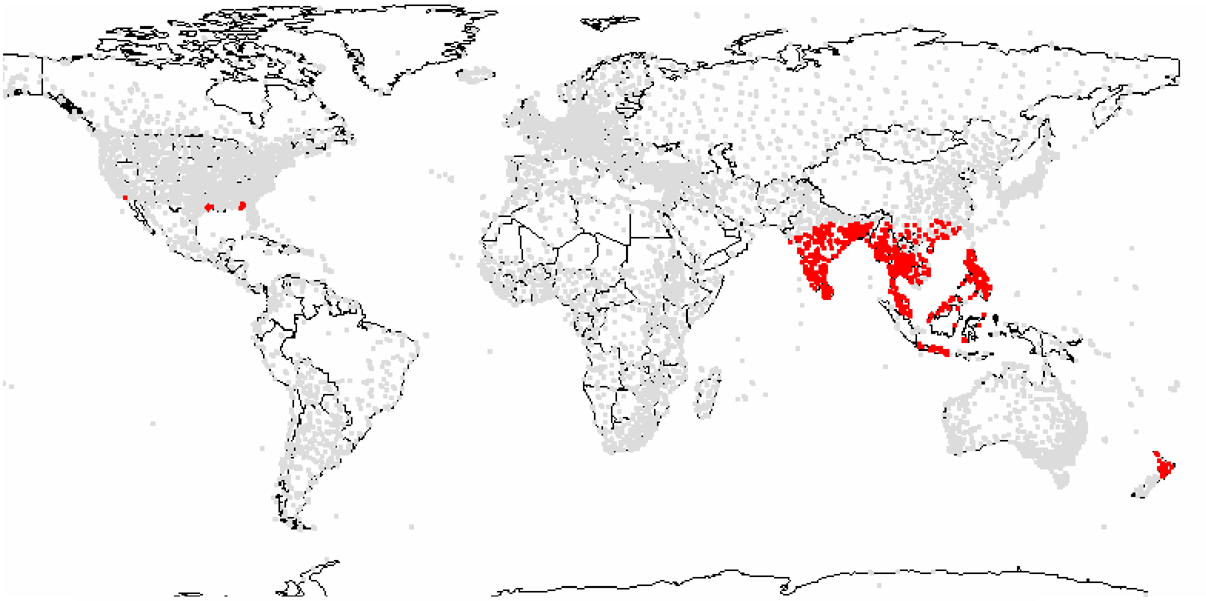


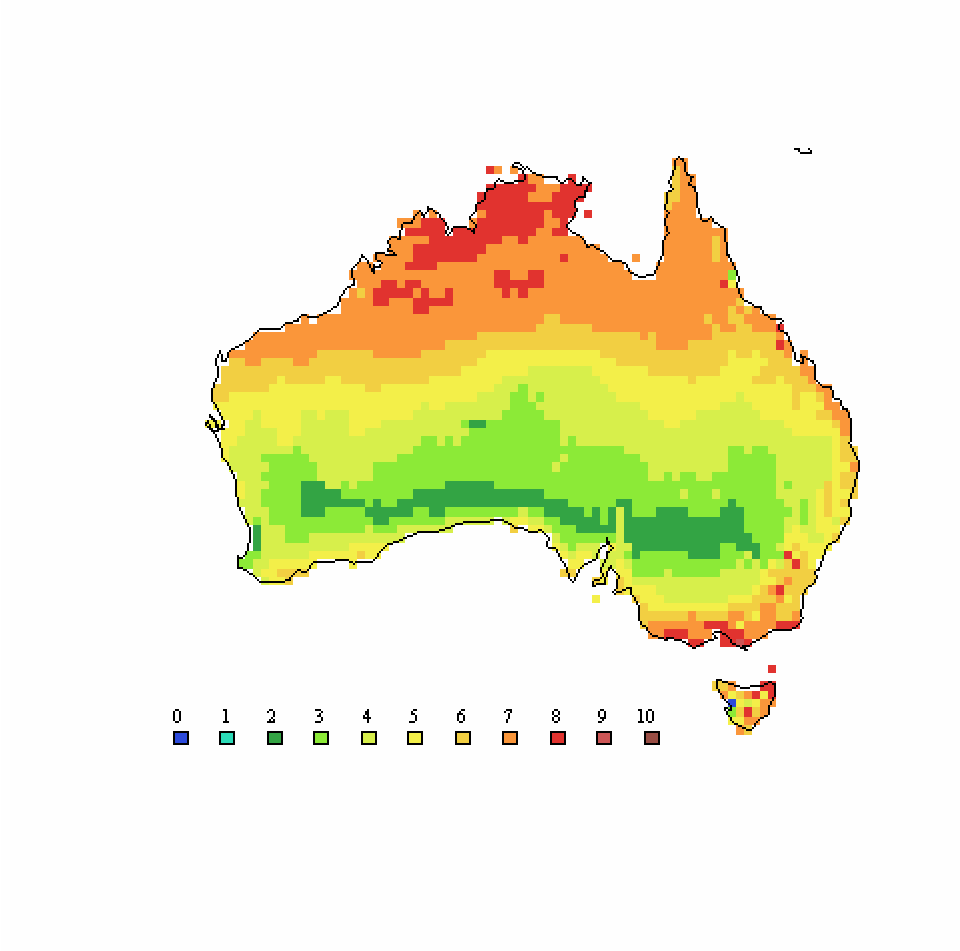
Figures 15a and 15b Hog deer *Cervus porcinus*



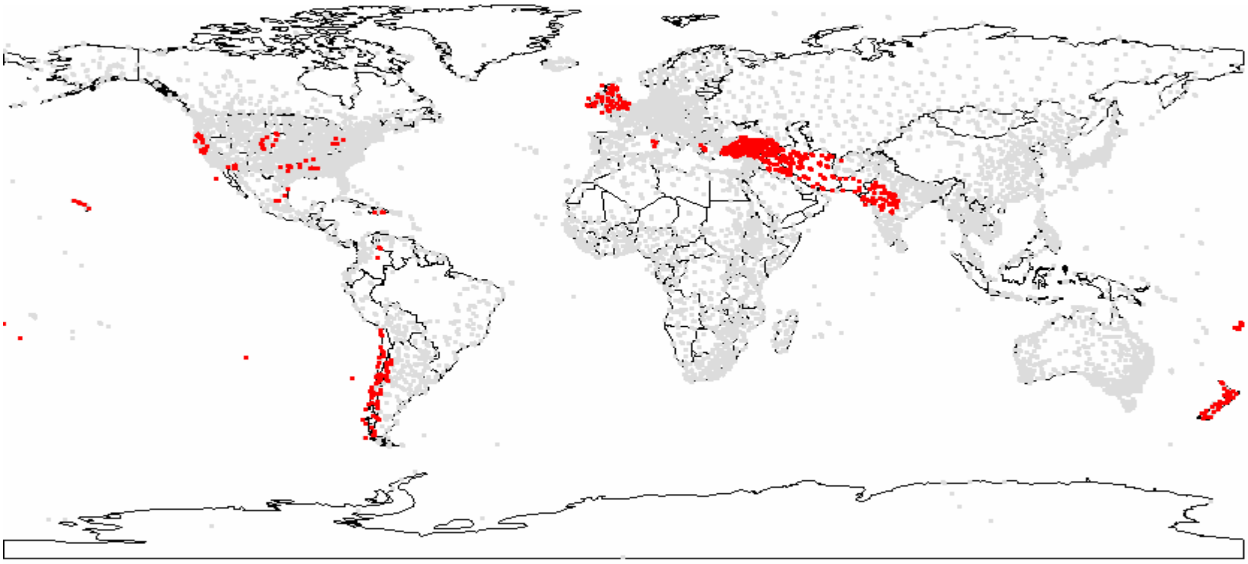


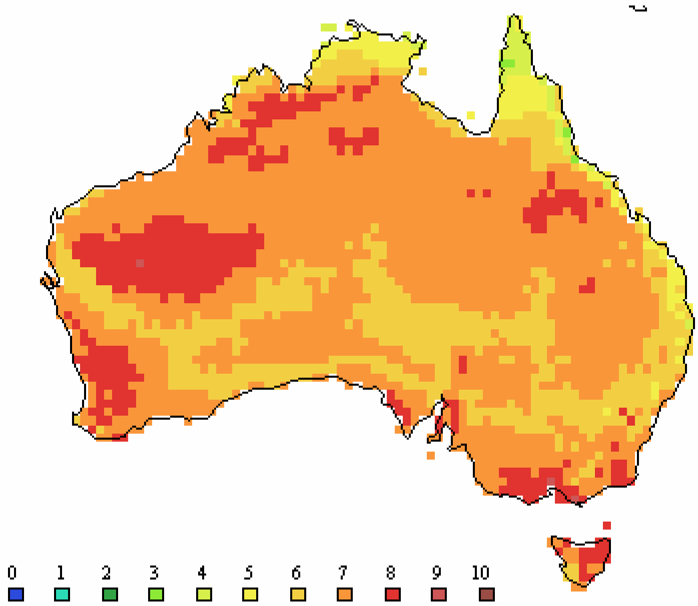
Figures 16a and 16b Timor or Rusa deer, *Cervus timorensis*



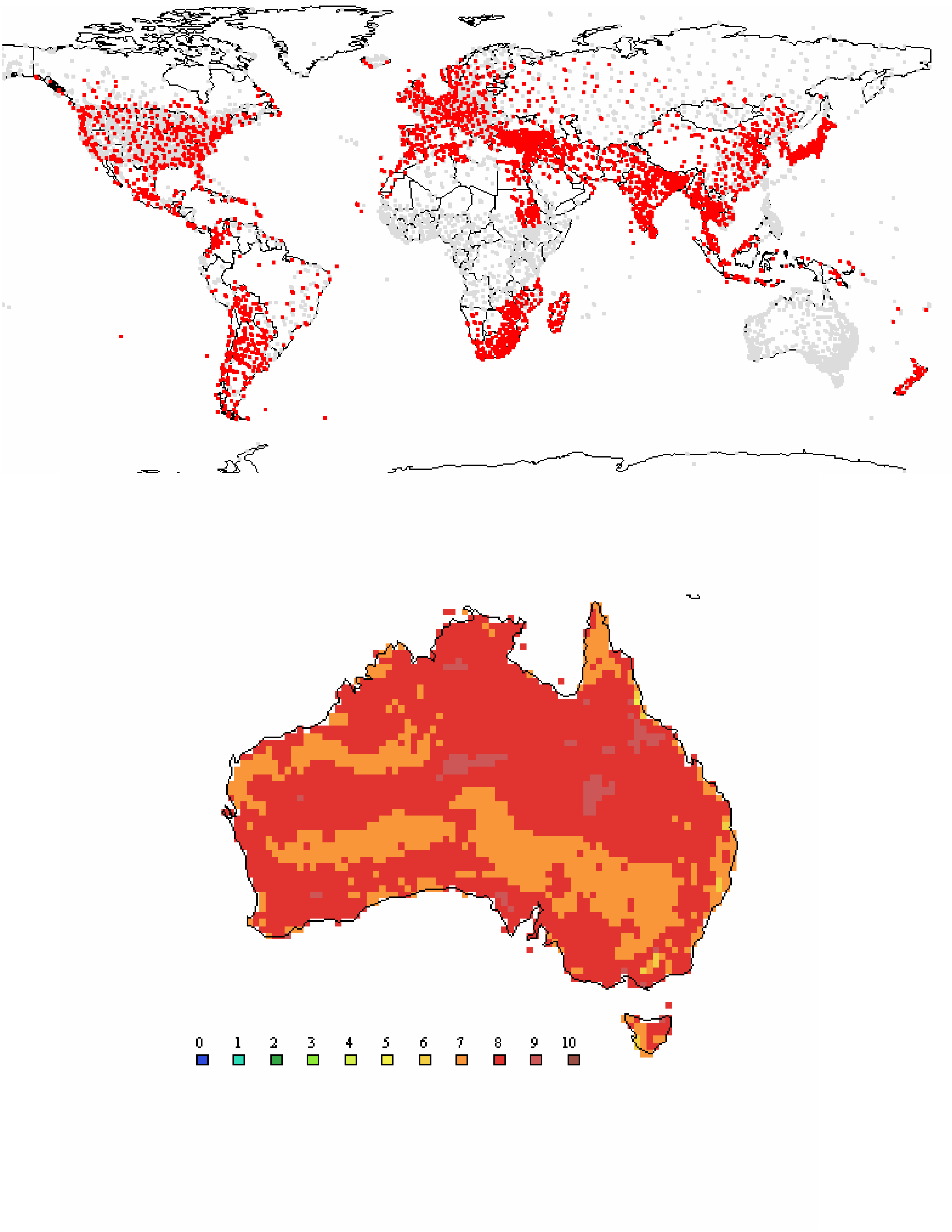


Figures 17a and 17b Sambar *Cervus unicolor*

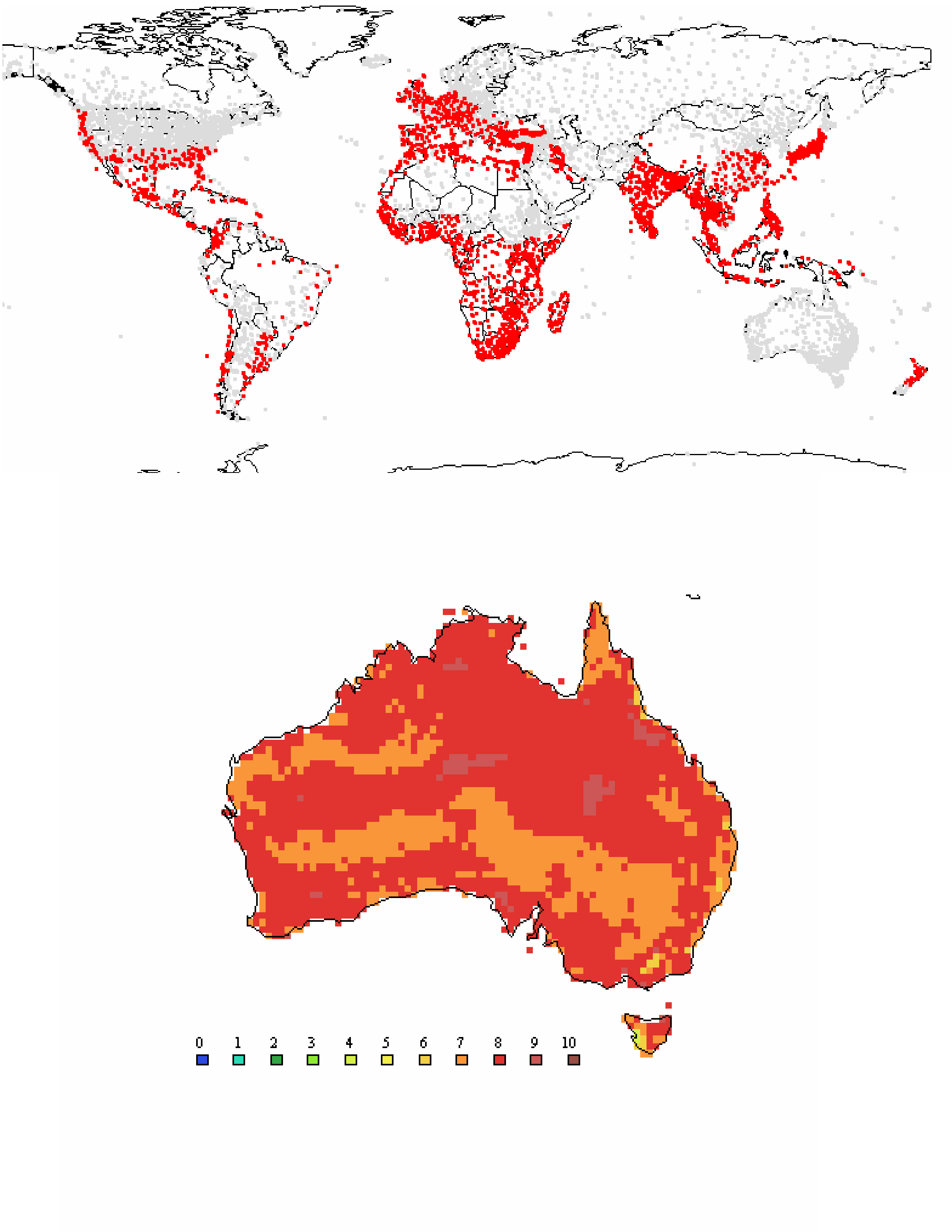




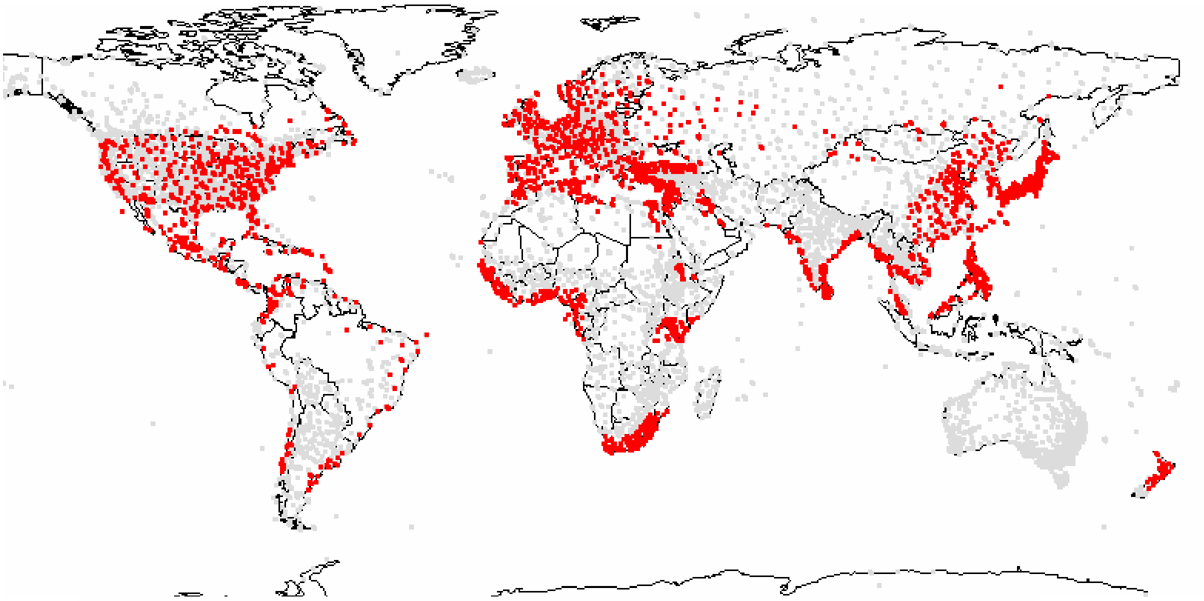
Figures 18a and 18b Feral goat *Capra hircus*

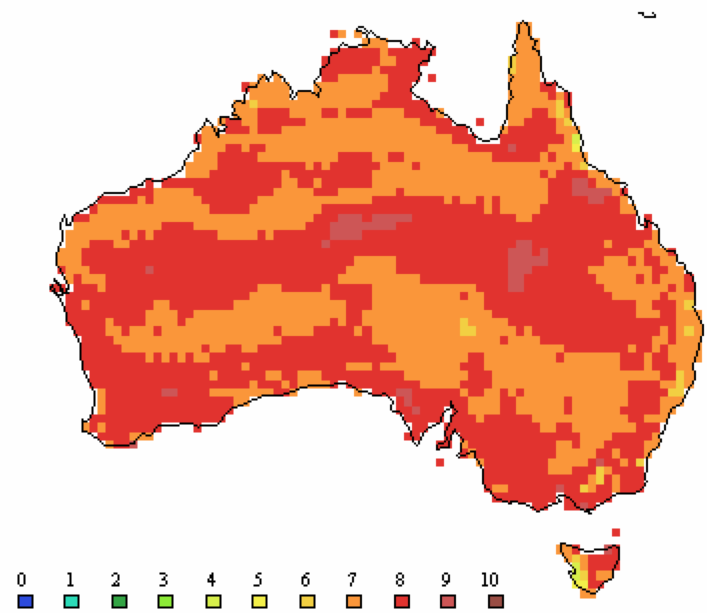


Figures 19a and 19b House mouse *Mus domesticus*

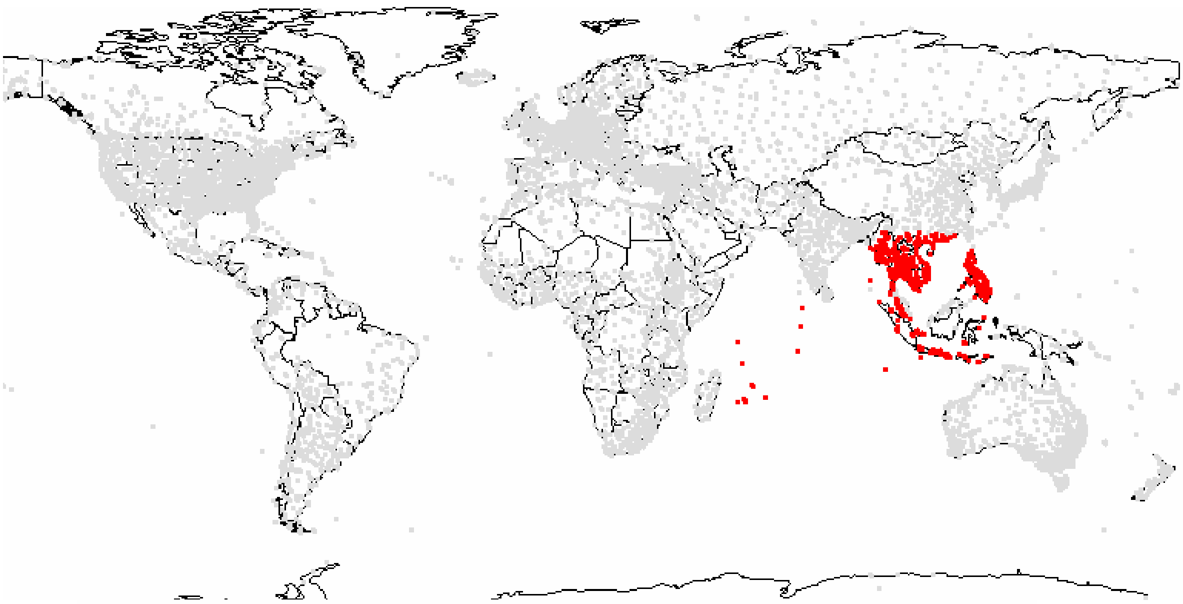


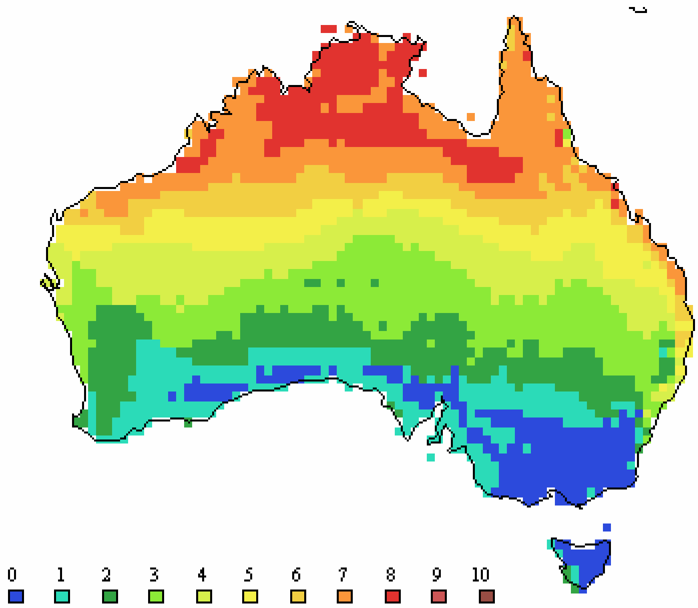
Figures 20a and 20b Black rat *Rattus rattus*



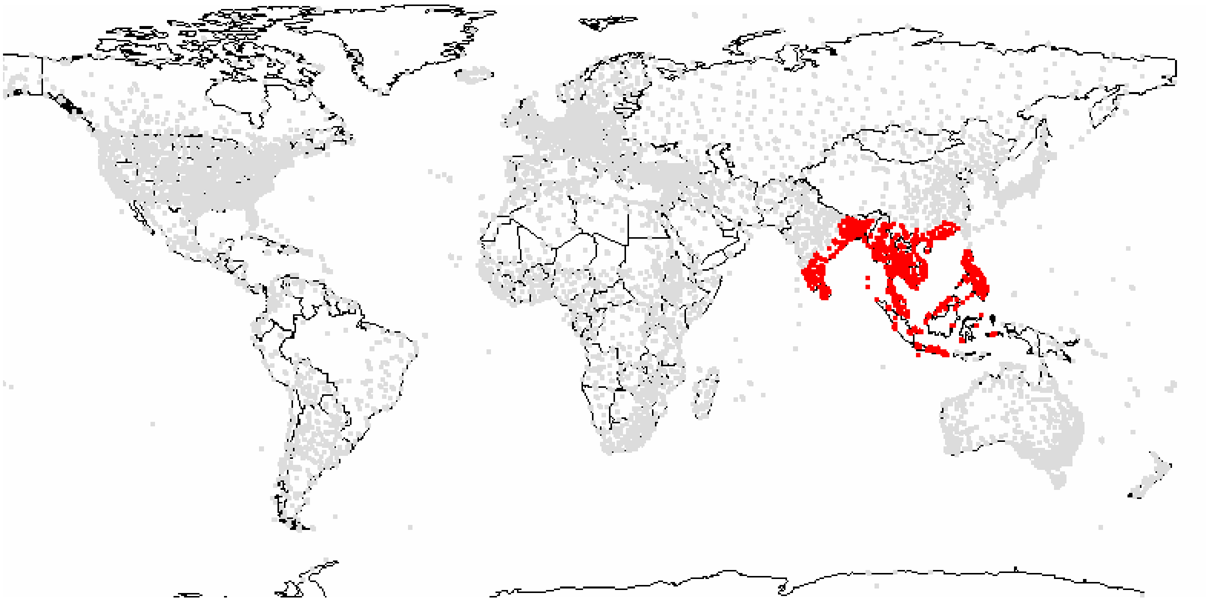


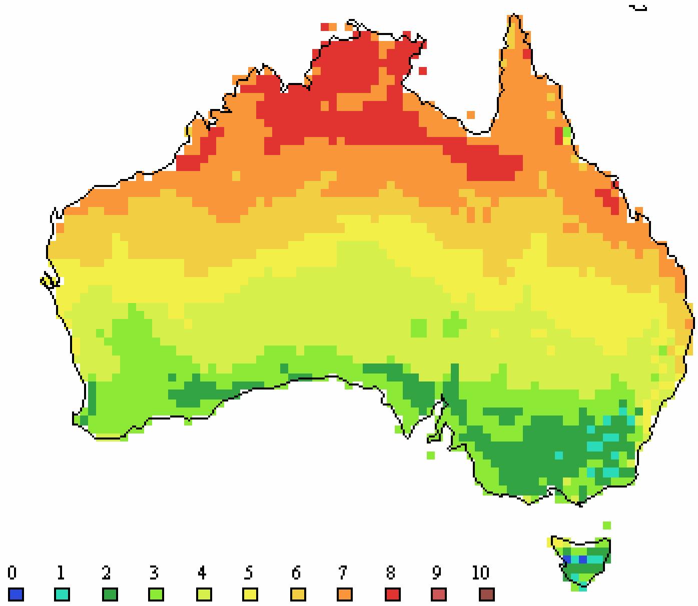
Figures 21a and 21b Brown rat *Rattus norvegicus*



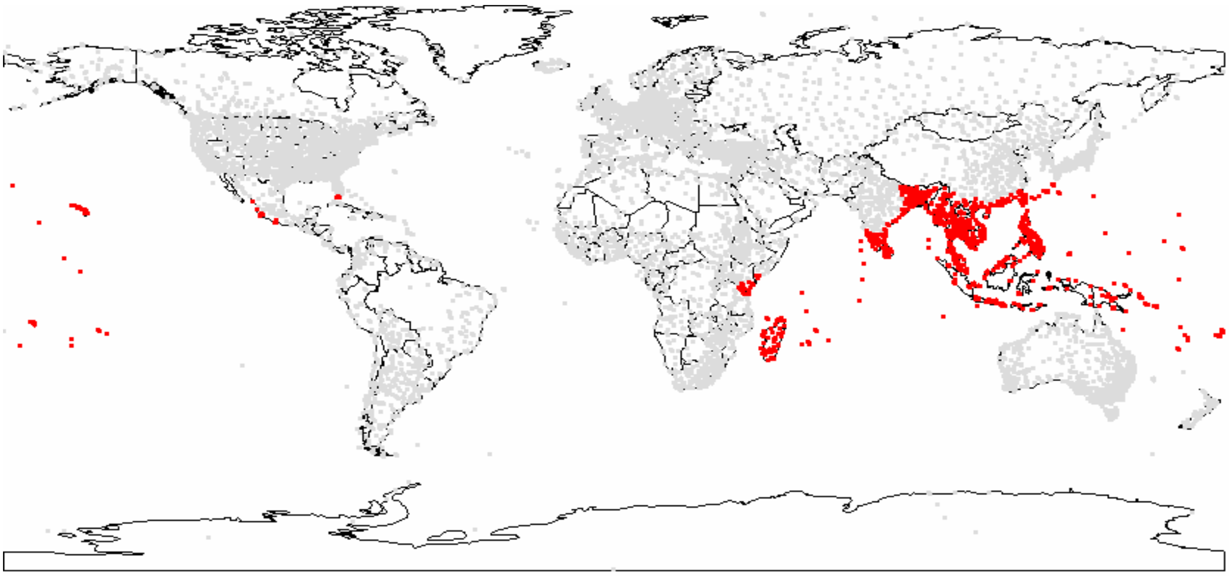


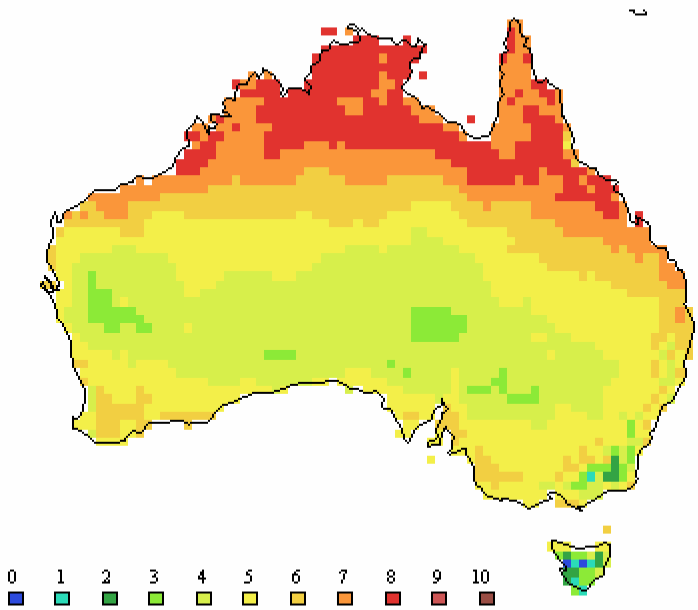
Figures 22a and 22b Wolf snake *Lycodon capucinus*



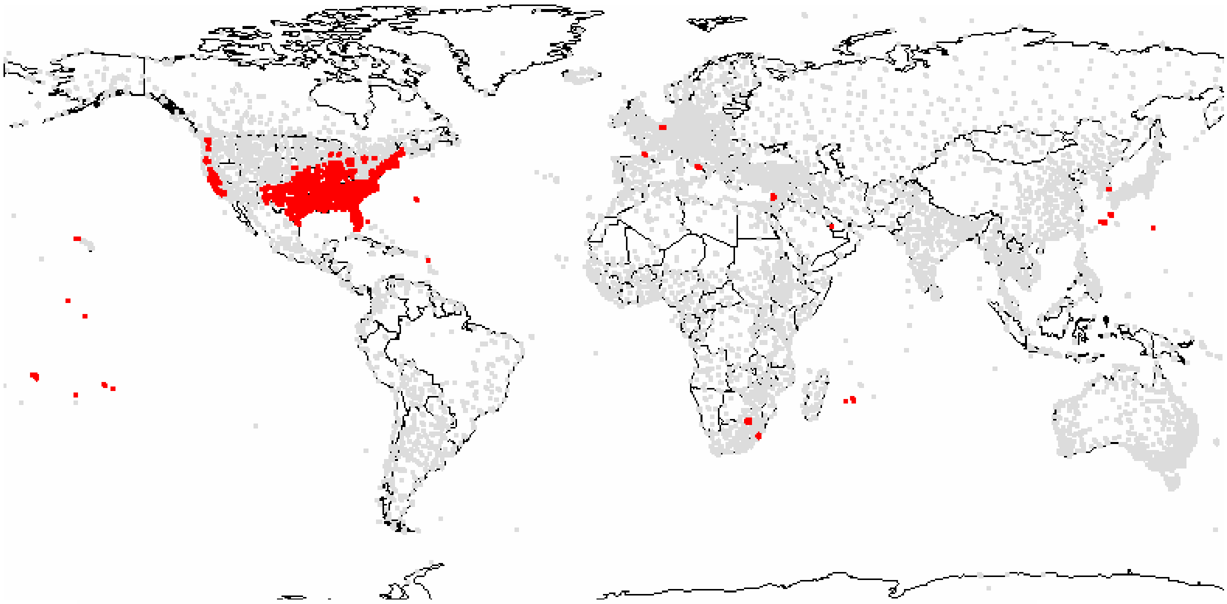


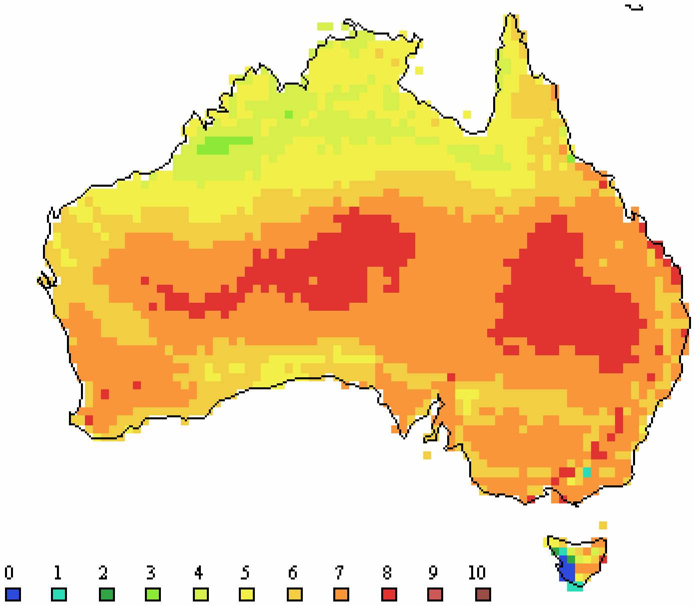
Figures 23a and 23b Grass skink *Lygosoma bowringii*





Figures 24a and 24b House gecko *Hemidactylus frenatus*





Figures 25a and 25b Red-eared slider *Trachemys scripta elegans*

Appendix 3. Criteria to Assess Whether Local Eradication is Possible

Eradication of established pest animals is possible only on a local scale. To determine whether eradication is likely to be successful, six criteria can be applied: three essential for the achievement of eradication and three to help managers decide whether eradication is preferable to ongoing control (Bomford and O'Brien, 1995).

Essential

- *Pests can be killed at a faster rate than they can replace themselves*

This seems obvious but it is difficult to achieve in practice. There are two main reasons. Firstly, many pest populations have a high natural rate of increase. Secondly, as the density of a pest declines, it takes progressively more time and more expense per individual animal to locate and remove the last few animals.

- *Immigration can be prevented*

This criterion can be met for small, isolated water bodies but is very difficult to achieve over a wide area. If animals can recolonise an area from nearby populations or by escape or release from captive populations such as domestic aquariums, elimination of the pest will at best be temporary. Immigration to a local area may be prevented where a suitable structure and control creates a perfect barrier.

- *All reproductive individuals are at risk from the available techniques*

It is not necessary to remove all pest animals at the first attempt. However, all reproductive or potentially reproductive members of the pest population must be able to be taken by the techniques available. This is rarely possible in part because there is only a limited armory of techniques. If, for example, some animals avoid poisoned baits then those animals cannot be removed and eradication will not be achieved. Trap-shyness and bait-avoidance, and resistance to poisons, are common among pest animals.

Desirable

- *The pest can be monitored at very low densities*

If the animal cannot be detected at very low densities, then there is no way of knowing whether all animals have been eliminated. However, most population assessment techniques cannot detect animals at very low densities. The difficulty in meeting this criterion is illustrated by the attempts to remove rabbits from Phillip Island off Norfolk Island. A small population of rabbits was found on the island two years after it was thought that all of them had been removed.

- *The socio-political environment supports eradication*

Even when all the technical problems can be met, social and political factors may prevent successful eradication. Community attitudes may oppose killing large numbers of animals on moral, emotional or cultural grounds. Also, eradication is expensive. Political factors may withdraw funds from the program before eradication is achieved.

- *The high costs of eradication can be justified.*

It is appealing to think that the value of perpetual freedom from a pest is very high, but this may not be so. Future benefits such as those obtained from eradicating pests have a lower economic value than benefits that are available immediately. This is because the value of future benefits is discounted. Calculating discount rates involves the reverse of the equation to calculate interest rates on invested money. Using a hypothetical model of the costs and benefits of eradication it was shown that when the discount rate was set at zero eradication became cost effective after 28 years. Setting a very low discount rate of 3.5% made eradication cost effective after 47 years, but, at 10%, eradication never became cost effective. The practice of discounting the value of future benefits assumes that land managers act in an economically rational manner. However, pests often evoke strong emotional responses to the extent that management aims and expenditure are often far from rational. The resource being protected also has to have a monetary value allocated to it in order to determine whether eradication is economic. Yet the monetary value of conservation and biodiversity is difficult to assess. There are methods to do so, such as contingent valuation, but their usefulness is debatable.