CHAPTER 17

Australian reptiles and their conservation

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Summary

Australia has a spectacular and diverse reptile fauna approaching 1000 species, 93% of which are endemic to the continent. Despite this, there is a paucity of information on the biology of Australian reptiles compared with mammals and birds. The single greatest threat to Australian reptiles is the removal of native vegetation, most of which has occurred in the state of Queensland during the past few decades. Since European settlement in Australia, land clearing for stock grazing and other agricultural activities has reduced the extent of native vegetation, and resulted in extensive habitat fragmentation. Ultimately, habitat fragmentation leads to species loss and local extinctions. Other threats to Australian reptiles include livestock grazing, which occurs on 55% of the continent, coupled with changing fire regimes and predation by exotic predators, especially foxes and feral cats. Currently, we know little about the long-term impacts of pastoralism, fire and introduced predators on reptile communities. The conservation of Australian reptiles requires urgent changes in government policy to reduce rates of vegetation clearing. A critical challenge is the conservation of reptiles in the vast arid and semi-arid regions, where reptile diversity is remarkably high. This will require coordinated management of threatening processes across multiple land tenures, including pastoral leases, crown lands, Aboriginal lands and conservation reserves. In southern Australia, the conservation of reptiles in fragmented landscapes will require strategic tree planting to increase the sizes of habitat remnants and their connectivity, in addition to retaining important structural habitat features such as rock outcrops, old growth trees and fallen timber. In addition to *in situ* conservation practices, breeding programmes are being employed to prevent the extinction of imperilled species.

17.1 Introducing Australia's reptiles

Australia has close to 1000 species of reptiles and at least 189 described subspecies, representing 18 families and 163 genera, which equates to almost 10% of the world's reptile fauna. Numerically, Australia has the most endemic species of any country, with 93% of its

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reptile species unique to the continent. The arid zone, which covers two-thirds of the continent, harbours one of the most diverse lizard assemblages on the planet. In the Great Victoria Desert, you can find 47 species of lizards living together at the same sand ridge site. No other deserts come close to matching this diversity; North American deserts harbour just 12 lizard species, while only 20 lizard species occur in the Kalahari Desert in southern Africa (Pianka, 1986). Roughly 6% of Australia's reptile species are threatened (Table 17.1), comprising seven species listed as critically endangered, 17 species listed as endangered, and 34 species listed as vulnerable (EPBC, 2013). Taxa with disproportionate numbers of threatened species include marine turtles (100%: six of six species), freshwater turtles (22%: five of 23 species), and pygopodid lizards (18%: seven of approximately 40 species). Why so many legless lizards are threatened is perplexing. This small family, endemic to Australia and New Guinea, displays an extraordinarily high diversity of diets and foraging modes.

Table 17.1 Threatened Australian reptiles (EPBC Act List of Threatened fauna 2014)

Group	Scientific Name	Classification
Sea Turtles		
Loggerhead Turtle	*Caretta caretta*	Endangered
Leatherback Turtle	*Dermochelys coriacea*	Endangered
Olive Ridley Turtle	*Lepidochelys olivacea*	Endangered
Hawksbill Turtle	*Eretmochelys imbricata*	Vulnerable
Green Turtle	*Chelonia mydas*	Vulnerable
Flatback Turtle	*Natator depressus*	Vulnerable
Freshwater Turtles		
Western Swamp Turtle	*Pseudemydura umbrina*	Critically Endangered
Gulf Snapping Turtle	*Elseya lavarackorum*	Endangered
Mary River Turtle	*Elusor macrurus*	Endangered
Bell's Turtle (Namoi River)	*Myuchelys bellii = Wollumbinia belli*	Vulnerable
Fitzroy River Turtle	*Rheodytes leukops*	Vulnerable
Snakes		
Short-nosed Seasnake	*Aipysurus apraefrontalis*	Critically Endangered
Leaf-scaled Seasnake	*Aipysurus foliosquama*	Critically Endangered
Plains Death Adder	*Acanthophis hawkei*	Vulnerable
Ornamental Snake	*Denisonia maculata*	Vulnerable
Dunmall's Snake	*Furina dunmalli*	Vulnerable
Broad-headed Snake	*Hoplocephalus bungaroides*	Vulnerable
Olive Python (Pilbara subspecies)	*Liasis olivaceus barroni*	Vulnerable
Krefft's Tiger Snake (Flinders Ranges)	*Notechis scutatus ater*	Vulnerable
Christmas Island Blind Snake	*Ramphotyphlops exocoeti*	Vulnerable
Lizards		
Nangur Spiny Skink	*Nangura spinosa = Concinnia spinosa*	Critically Endangered
Christmas Island Blue-tailed Skink	*Cryptoblepharus egeria*	Critically Endangered
Christmas Island Forest Skink	*Emoia nativitatis*	Critically Endangered
Lister's Gecko (Christmas Island)	*Lepidodactylus listeri*	Critically Endangered

Table 17.1 *(cont.)*

Group	Scientific Name	Classification
Christmas Island Giant Gecko	*Cyrtodactylus sadleiri*	Endangered
Arnhem Land Egernia	*Bellatorias obiri*	Endangered
Alpine She-oak Skink	*Cyclodomorphus praealtus*	Endangered
Baudin Island Spiny-tailed Skink	*Egernia stokesii badia*	Endangered
Blue Mountains Water Skink	*Eulamprus leuraensis*	Endangered
Corangamite Water Skink	*Eulamprus tympanum marnieae*	Endangered
Allan's Lerista	*Lerista allanae*	Endangered
Guthega Skink	*Liopholis guthega*	Endangered
Slater's Skink	*Liopholis slateri slateri*	Endangered
Yellow-snouted Gecko	*Lucasium occultum*	Endangered
Adelaide Bluetongue Lizard	*Tiliqua adelaidensis*	Endangered
Grassland Earless Dragon	*Tympanocryptis pinguicolla*	Endangered
Five-clawed Worm-skink	*Anomalopus mackayi*	Vulnerable
Pink-tailed Legless Lizard	*Aprasia parapulchella*	Vulnerable
Flinders Ranges Worm-lizard	*Aprasia pseudopulchella*	Vulnerable
Hermite Island Worm-lizard	*Aprasia rostrata rostrata*	Vulnerable
Lord Howe Island Gecko	*Christinus guentheri*	Vulnerable
Three-toed Snake-tooth Skink	*Coeranoscincus reticulatus*	Vulnerable
Yinnietharra Rock-Dragon	*Ctenophorus yinnietharra*	Vulnerable
Airlie Island Ctenotus	*Ctenotus angusticeps*	Vulnerable
Lancelin Island Skink	*Ctenotus lancelini*	Vulnerable
Hamelin Ctenotus	*Ctenotus zastictus*	Vulnerable
Striped Legless Lizard	*Delma impar*	Vulnerable
Atherton Delma	*Delma mitella*	Vulnerable
Collared Delma	*Delma torquata*	Vulnerable
Yakka Skink	*Egernia rugosa*	Vulnerable
Houtman Abrolhos Spiny-tailed Skink	*Egernia stokesii aethiops = Egernia stokesii badia*	Vulnerable
Mount Cooper Striped Lerista	*Lerista vittata*	Vulnerable
Great Desert Skink	*Liopholis kintorei*	Vulnerable
Jurien Bay Skink	*Liopholis pulchra longicauda*	Vulnerable
Pedra Branca Skink	*Niveoscincus palfreymani*	Vulnerable
Lord Howe Island Skink	*Oligosoma lichenigera*	Vulnerable
Bronzeback Snake-lizard	*Ophidiocephalus taeniatus*	Vulnerable
Granite Belt Thick-tailed Gecko	*Uvidicolus sphyrurus*	Vulnerable

Some small worm-like fossorial species gorge themselves on ant pupae and larvae, much like blind snakes; some diurnally active foragers feed mostly on spiders; while others feed on insects. One species, Burton's snake lizard, is a snake analogue with specialised hinged teeth that allows it to subdue scincid lizards, which it ambushes from leaf litter (Greer, 1997). If space permitted, we could go on describing the fascinating and remarkable reptiles in Australia, but this is not our aim. Instead, we highlight the major threatening processes which endanger Australian reptiles, and offer some potential solutions.

17.2 Threatening processes and mitigating actions

17.2.1 Removal of native vegetation

Clearing of native vegetation poses the single greatest threat to Australian reptiles (Cogger *et al.*, 1993, 2003). Despite the well documented problems associated with broad-scale vegetation removal, such as soil erosion, hydrological changes, and dry land salinity (Taylor & Hoxley, 2003), Australia has one of the highest rates of native vegetation clearing in the world. Incredibly, more vegetation has been removed in recent decades than at any other time in Australia's history (Bradshaw, 2012). Since 1988, most vegetation clearing has occurred in the state of Queensland, where >6 million ha of native vegetation have been felled for livestock grazing and other agricultural activities (Bradshaw, 2012, DERM, 2010). Rates of land clearing in QLD remained high until 2008; approximately 700 000 ha were cleared annually between 1988 and 1990, while over 300 000 ha were cleared annually between 1990 and 2006 (DERM, 2010). This fell to 99 000 ha per year in the period 2008–2009, but the recent relaxation of vegetation clearing laws by the Queensland government may lead to a resurgence of vegetation removal. These losses are particularly troubling when one considers that Queensland supports over half of Australia's terrestrial endemic reptile species, and is a recognised hotspot for diverse reptile groups including geckos, skinks and snakes (Cogger *et al.*, 1993).

Vegetation removal affects reptiles via direct and indirect pathways, which operate over short and long timescales. In the short term, many reptiles are killed or receive life-threatening injuries from clearing activities (Cogger *et al.*, 2003). Following the removal of vegetation, many surviving reptiles will likely be killed by aerial predators (raptors, corvids, kookaburras, owls) or feral cats, foxes and dingoes. Although some reptiles may find some temporary shelter in woodpiles, most of these animals will die when the piles are subsequently burnt. Reptile deaths caused by vegetation clearing are staggering; during the period 1997–1999 it was estimated that 89 million individuals were killed in Queensland each year (Cogger *et al.*, 2003). Such losses cannot be replaced once the habitat is destroyed; indeed, you only have to walk through a treeless paddock to see that very few reptiles persist in these desolate landscapes (Driscoll, 2004).

Over longer timescales, local extinctions will continue to occur in any remaining habitat remnants (Tilman *et al.*, 1994). Such extinctions will occur due to multiple factors including edge effects, stochastic events (such as wildfires), mortality from motor vehicles, habitat degradation from livestock grazing, predation, and the inability of some species to use or recolonise the remnant patches (Driscoll, 2004). Ultimately, the effects of land clearing in Queensland will not be fully realised for several decades, and unless we take actions to curtail current rates of clearing, many endemic reptile species will likely be extinct in the next 30 years.

17.2.2 Habitat fragmentation

Habitat fragmentation, the end result of vegetation clearing, poses a serious threat to Australian reptiles (Cogger *et al.*, 1993, 2003). Indeed, habitat fragmentation was believed to have caused the extinction of the pygmy bluetongue lizard until it was rediscovered in a small habitat remnant in 1992 (Box 17.1).

Reptiles are particularly sensitive to habitat fragmentation due to their poor dispersal abilities (Williams *et al.*, 2012). Some reptiles rarely disperse across cleared areas, which in

Box 17.1 Rediscovery and conservation of the pygmy bluetongue lizard *Tiliqua adelaidensis*

The pygmy bluetongue lizard, *Tiliqua adelaidensis*, is the smallest (to 20 cm long) member of the skink genus *Tiliqua*, and was once widely distributed in South Australia. Habitat loss and fragmentation decimated populations, and the lizard was thought to be extinct until it was rediscovered near Burra in South Australia in 1992. Incredibly, two South Australian herpetologists found a pygmy bluetongue inside the stomach of a dead road-killed brown snake! Searches in the area eventually led to the discovery of live specimens (Armstrong *et al.*, 1993). The species currently occurs on just 31 disjunct sites in a small farming region of South Australia (Duffy *et al.*, 2012).

Pygmy bluetongue lizards inhabit modified grasslands (dominated by exotic grasses), native grasslands, and grassy woodlands (Souter *et al.*, 2007). The lizards use empty burrows of wolf (lycosid) and trapdoor (mygalomorph) spiders as shelters, basking sites and ambush foraging sites (Milne & Bull, 2000, Hutchinson *et al.*, 1994). Population density ranges from 15 to 200 individuals per hectare, and is highly variable across sites (Duffy *et al.*, 2012). Female pygmy bluetongues are viviparous, and give birth to one to four young between January and March. Juveniles disperse to unoccupied burrows after birth, but fewer than 10% of juveniles survive to maturity (Milne, 1999). Pygmy bluetongues mature early (around one to two years) and once mature, adults seldom move further than 20 m from their burrows (Milne, 1999). These life-history traits, coupled with habitat fragmentation, may explain why gene flow is restricted both within and between populations (Smith *et al.*, 2009).

Threats to pygmy bluetongue lizards

Only 0.3% of the original native grasslands within the pygmy bluetongue's historical range remain; the remainder has been cleared and fragmented (Hyde, 1995). Extant populations of pygmy bluetongue lizards occur on private land and are threatened by inappropriate grazing and agricultural activities that disturb the soil. Ploughing and ripping of the soil can kill or injure lizards, and destroys the spider burrows that are crucial habitat requirements (Duffy *et al.*, 2012). Grazing helps to maintain basking sites around burrows and may prevent weed invasions, but overstocking sensitive grassland habitats could reduce prey availability and damage spider burrows (Souter *et al.*, 2007). Planting trees in grasslands could also provide roosting and nesting sites for birds, thereby increasing avian predation rates on lizards (Duffy *et al.*, 2012). Survival is a balancing act at many levels for the pygmy bluetongue lizard.

Because pygmy bluetongue lizards have specific habitat requirements, and extant populations are small and isolated, this species is particularly vulnerable to climate change. Low rates of gene flow in extant populations suggests that the species would have difficulty dispersing to new habitats should the current area of occupancy become unsuitable (Fordham *et al.*, 2012). Changes to fire regimes could also threaten extant lizard populations. For example, intense grass fires during the juvenile dispersal phase could increase juvenile mortality rates, which could reduce local populations or reduce dispersal rates across fragmented landscapes (Fenner & Bull, 2007).

Provision of burrows and mitigation of current threats

The future of the pygmy bluetongue lizard will depend on how well we can protect and maintain existing habitats and populations. Ultimately, recovery of this species will require working closely with landholders to actively manage grazing lands (Duffy *et al.*, 2012). Strategic rotational grazing may be necessary to maintain habitat quality (Clarke, 2000), and activities that disturb the soil will need to avoid areas occupied by lizards. Artificial burrows are a promising avenue for enhancing existing populations and establishing suitable reintroduction sites to increase the number of extant populations (Souter *et al.*, 2004). Increased burrow densities could reduce juvenile mortality rates and help increase lizard densities at established sites (Souter *et al.*, 2004). Although there are several challenges to conserving pygmy bluetongue lizards, with the cooperation of landholders, researchers and conservation groups, their future looks secure.

turn can alter the genetic structure of populations within remnants (Stow *et al.*, 2001). Over time, isolated populations may suffer from inbreeding or loss of genetic diversity, further increasing their vulnerability to extinction (Frankham, 2005). Species with specialised habitat requirements and/or small geographic ranges, such as the endangered broad-headed snake (Box 17.2) and endangered Nangur spiny skink (Box 17.3) are particularly sensitive to habitat fragmentation. However, fragmentation also affects habitat generalists. Many reptiles require leaf litter, fallen timber and rocks for shelter, thermoregulation and/or foraging, and the removal of fallen timber or paddock trees for firewood, coupled with livestock grazing, and the invasion of weeds, can degrade the quality of habitat patches over time (Cunningham *et al.*, 2007; Dorrough *et al.*, 2012). Hence, through chance events, small isolated reptile populations in an agricultural matrix may be on a path to extinction even if no further habitat loss occurs (Tilman *et al.*, 1994).

Studies in extensively cleared agricultural areas of southern Australia, where >90% of the original vegetation has been cleared, paint a particularly bleak future for reptiles. In gimlet *Eucalyptus salubris* woodland in the Western Australian wheatbelt, smaller remnants contained fewer reptile species than larger remnants (Kitchener & How, 1982; Smith *et al.*, 1996). In general, woodland remnants had a depauperate lizard fauna that was dominated by generalist species (Smith *et al.*, 1996). Driscoll (2004) found that the painted dragon *Ctenophorus pictus* and the hooded scalyfoot *Pygopus nigriceps* were locally extinct in habitat remnants in south-western NSW. An even more depressing situation was recorded by Brown *et al.* (2008). These authors sampled reptile assemblages in habitat remnants in the Victorian Riverina district, and found no reptiles at 22% of sites! Moreover, over half of the reptiles they observed were two common, widespread, generalist skinks (Brown *et al.*, 2008). Collectively, these results suggest that regional reptile extinctions have already occurred in fragmented agricultural landscapes.

Box 17.2 Restoring habitats for the broad-headed snake *Hoplocephalus bungaroides*

The broad-headed snake *Hoplocephalus bungaroides* is a small (to 90 cm snout–vent length), spectacularly coloured nocturnal elapid snake (Figure 17.1, Plate 34). The species is restricted to sandstone rock formations within a 200 km radius of Sydney, Australia's largest city (Cogger *et al.*, 1993). During the cooler months, broad-headed snakes thermoregulate underneath sun-exposed sandstone rocks or inside crevices, and during summer they shelter in tree hollows (Webb & Shine, 1998). In 1850, broad-headed snakes were common throughout the Sydney region, but by 1869 the species was becoming scarce due to the removal of 'bush rock' by builders and gardeners (Krefft, 1869). Today, the species is locally extinct in the Sydney metropolitan area, and is confined to a handful of disjunct populations south, west and north of Sydney.

Broad-headed snakes grow slowly, mature late, are long lived, and females reproduce infrequently. The snakes' slow life history, low juvenile dispersal and habitat specificity make it particularly vulnerable to extinction (Webb *et al.*, 2002b). Current threats include the removal of snakes for the illegal pet trade, the removal of sandstone rocks for supply to nurseries (below), the destruction of habitat associated with illegal reptile collecting activities, and overgrowth of rock outcrops by emergent vegetation (Webb *et al.*, 2002a; Pringle *et al.*, 2009; Pike *et al.*, 2010). To mitigate some of these threats, National Parks and Wildlife staff implemented a series of management actions including the erection of locked gates to exclude vehicular access to some populations (Figure 17.2, Plate 55), signage to inform the public that bush rock collection is illegal, and the installation of hidden remotely triggered cameras to record the number plates of vehicles used by snake collectors or bush rock collectors. Nonetheless, collectors continue to damage gates to gain access to such sites (Figure 17.2, Plate 55), suggesting that broad-headed snakes are prized by collectors.

Figure 17.1 (Plate 34) The broad-headed snake is highly prized by snake enthusiasts due to its rarity and beautiful colouration. Photograph by Jonathan Webb. A black and white version of this figure will appear in some formats. For the colour version, please refer to the plate section.

Restoration of rock outcrops degraded by bush rock collectors

One of the major threats to broad-headed snakes is the removal of 'bush rocks' for landscaping urban gardens. Bush rock removal is listed as a key threatening process under the NSW Threatened Species Conservation Act 1995 yet, incredibly, there is no legislation outlawing the collection or sale of bush rocks! Many rock outcrops in the Sydney region were stripped of their surface rocks during the 1960s and 1970s, and consequently, these degraded sites have few suitable shelters for broad-headed snakes or other rock-dwelling reptiles.

To restore degraded rock outcrops, researchers developed fibre-reinforced cement rocks with thermal attributes that mimic those of sandstone rocks favoured by broad-headed snakes (Croak *et al.*, 2010). These artificial rocks were used to restore degraded rock outcrops at several locations in the Sydney region. Encouragingly, velvet geckos, a major prey of juvenile broad-headed snakes, colonised the artificial rocks within months of deployment (Croak *et al.*, 2010). Just one year later, broad-headed snakes began using the rocks (Croak *et al.*, 2012). Future restoration of degraded rock outcrops, coupled with the translocation of juvenile broad-headed snakes to restored sites, could help to prevent the extinction of this iconic elapid snake.

Figure 17.2 (Plate 55) In some cases it may be necessary to restrict access to endangered reptile populations. Gates were erected to prevent vehicular access to populations of broad-headed snakes (*Hoplocephalus bungaroides*) and Nangur spiny skinks (*Nangura spinosa*). Although gates can receive frequent vandalism, if maintained they can deter human access. In the left-hand photo, the gate has been pulled out of the ground, despite being held in place with concrete. In the right-hand photo the gate has been pulled off the hinges, and is hanging open by the locked chain. These types of activities make conserving endangered reptiles more challenging. Photographs by David Pike. A black and white version of this figure will appear in some formats. For the colour version, please refer to the plate section.

17.2.2.1 Revegetation of fragmented agricultural landscapes

Australia has 42 highly fragmented subregions containing less than 30% of the original vegetation. These subregions occur in south-western Western Australia, south-eastern South Australia, central and western Victoria, the New England Tablelands of New South Wales and the southern and central parts of eastern Queensland. The vegetation types most affected by vegetation removal are eucalypt woodlands, eucalypt open forests, and mallee woodlands and shrublands (Bradshaw, 2012). To conserve reptiles in fragmented agricultural landscapes, we urgently need to revegetate and protect

Box 17.3 An uncertain future for the critically endangered Nangur spiny skink *Nangura spinosa*

The Nangur spiny skink, *Nangura spinosa*, occurs only in Nangur National Park, west of Gympie in south eastern Queensland (Borsboom *et al.*, 2010). The two known populations support roughly 45 and 140 individuals in areas <8 ha and 868 ha in size, respectively (Borsboom *et al.*, 2010). These small disjunct populations are extremely vulnerable to extinction. Conserving such a narrowly distributed endemic species should in theory be straightforward; simply protect the known habitat from threats that could reduce population size or hinder population growth. In the case of the Nangur spiny skink, however, this problem is more complex; this species is vulnerable to many threats facing small populations *and* some of those that threaten widely distributed species. Local threats include poaching by collectors (Borsboom, 2012) and genetic bottlenecks due to inbreeding (Borsboom *et al.*, 2010). Widespread threats include habitat fragmentation and degradation (e.g. selective logging) and negative effects of invasive species (both animals and plants (Borsboom *et al.*, 2010, Borsboom, 2012)). Whether these threats combined will push Nangur spiny skinks to extinction is unknown, mainly due to a lack of the general ecological information necessary to adequately guide conservation and management efforts.

existing remnants (Driscoll, 2004). Thankfully, recent community-sponsored and government-funded initiatives have begun to protect vegetation remnants on some private lands, and develop replanting schemes and corridor plans for many areas of southern Australia.

Ideally, replanting projects should aim to enlarge and join existing forest remnants to maintain reptile species diversity (Driscoll, 2004). However, many reptile species require complex structural habitat features, such as logs, dead trees, large hollow-bearing trees, leaf litter, or rocks, which are often absent in newly revegetated habitat patches (Munro *et al.*, 2007). To conserve reptiles in agricultural landscapes, we need to retain these structural habitat components (Michael *et al.*, 2011). In areas where structural features are absent, the addition of coarse woody debris can enhance habitat suitability for reptiles, and may reduce the long time lag for the natural formation of such habitat features (Manning *et al.*, 2013). These conservation actions will require active participation by private landholders; in many states, private lands contain a significant proportion of remnant vegetation (Brown *et al.*, 2011b). Ultimately, we need to educate landholders about the benefits of retaining structural habitat features for native fauna if we are to conserve reptiles in fragmented agricultural landscapes.

17.2.3 The spread of the cane toad

The highly toxic cane toad *Rhinella marina* was introduced to Queensland in 1935 and has since spread across northern Australia. Many Australian reptiles lack physiological mechanisms to detoxify toad toxins, and die after mouthing or ingesting cane toads. Since cane toads invaded the Northern Territory, there have been massive declines in populations of varanid lizards (Doody *et al.*, 2009), freshwater crocodiles (Letnic *et al.*,

2008), and bluetongue lizards (Brown *et al.*, 2011a). As pointed out by Shine & Phillips (Chapter 5, this volume), some species may actually benefit from the arrival of cane toads because they no longer experience predation from varanid lizards. For example, in the Daly River region, green tree snakes increased in abundance after cane toads decimated populations of three species of varanid lizards (Doody *et al.*, 2013). Indirect effects of cane toads on reptiles could be positive or negative; for example, the removal of large varanid lizards might increase feral cat abundance, which, given the current declines of small mammals in northern Australia (Woinarski *et al.*, 2011) might increase predation on lizards and snakes. At present, we know little about the magnitude of these indirect effects of cane toads on reptiles (Shine & Phillips, Chapter 5, this volume).

Cane toads recently invaded the Kimberley region of Western Australia, which is recognised for its high reptile diversity and endemism (Cogger, 2000). The spread of cane toads will likely cause serious population declines of varanid lizards, some snakes and bluetongue lizards. However, given that we know virtually nothing about the interplay of cane toads with other threatening processes (predation, fire, and grazing), we hesitate to make any predictions about how cane toads may affect ecosystems in Western Australia.

17.2.3.1 Mitigating cane toad impacts

Despite much research on cane toads, we doubt that a method for eradicating cane toads will be developed in the foreseeable future. Traps baited with cane toad toxins will be useful for removing toad tadpoles from farm dams (Shine & Phillips, Chapter 5, this volume), but are impractical for reducing toad densities at a landscape scale. Encouragingly, replacing earthen farm dams with plastic water tanks could prevent cane toads from colonising semi-arid regions of the continent (Florance *et al.*, 2011). In fact, landscape-scale modelling demonstrated that strategic replacement of just 100 earthen dams with water tanks could prevent cane toads from reaching the Pilbara region of Western Australia (Tingley *et al.*, 2013). It would probably cost \$400 000 to keep cane toads out of the Pilbara (assuming poly tanks cost \$4000 each); this amount is trivial compared to the >\$1 million that has been squandered by community groups trying to eradicate cane toads via hand collection.

17.2.4 Changing fire regimes in tropical savannas and its impact on reptiles

Northern Australia is dominated by highly flammable tropical savannas which cover 1.9 million km^2. Temperatures are high year round, but most of the annual rainfall (400–1200 mm) falls in the four-month wet season (December–March), which is followed by an extended dry season. Prior to European settlement, savannas were populated by Aboriginal peoples who used fire for signalling, hunting, clearing country, and for promoting the growth of bush foods and vegetation that would attract macropods and other important prey species (Bowman, 1998). Most fires were lit early in the dry season (April–May), which created a mosaic of burnt and unburnt areas which prevented the spread of large, destructive late dry season fires. The loss of Aboriginal burning from this landscape in the 1960s resulted in a temporal shift to mid-to-late dry season fires that often burnt large tracts of savanna (Russell-Smith *et al.*, 2003). These regular, extensive fires have led to declines in flora and fauna across much of northern Australia (Vigilante & Bowman, 2004; Woinarski *et al.*, 2010).

Late dry season fires can cause direct mortality in frill-necked lizards, *Chlamydosaurus kingii* (Griffiths & Christian, 1996), while the removal of cover may increase the vulnerability of diurnal lizards to predation (Legge *et al.*, 2008). The major conservation challenge in savanna landscapes is to implement early dry season fires in a patchy manner across land tenures. Two exemplary projects in northern Australia – the Ecofire project (see Woinarski *et al.*, Chapter 25, this volume) and West Arnhem Land Fire Abatement (WALFA) project – demonstrate that this approach is not only possible, but can also yield significant biodiversity benefits. Importantly, such projects can empower Aboriginal people living in remote regions with few employment opportunities (Whitehead *et al.*, 2008). The WALFA project involves Aboriginal landowners and scientists working together to plan and manage fire on Aboriginal lands, using both on ground and aerial incendiaries to create fire breaks across the landscape. A recent study using Landsat imagery showed that the WALFA project has been highly successful; since the project was implemented, there has been a significant reduction in late dry season fires (from 29% to 12.5%) accompanied by a reduction in the mean annual proportion of country burnt from 38% to 30% (Price *et al.*, 2012).

17.2.5 Changing fire regimes in Australian deserts

The deserts of central Australia are dominated by fire-prone spinifex (*Triodia* spp.) landscapes, which contain a remarkably high diversity of reptiles (Pianka, 1986). The Aboriginal inhabitants of these landscapes used fire throughout the year for clearing country, signalling, hunting lizards, and for promoting the growth of bush foods. The movement of Aboriginal peoples from traditional lands to towns in Australian deserts during the 1960s resulted in a rapid shift from a patchwork mosaic of vegetation of different ages, to large patches of either long unburnt or recently burnt country (Burrows *et al.*, 2006). Current fire regimes are characterised by pulses of large, rainfall-driven wildfires which homogenise vast tracts of country (Edwards *et al.*, 2008). The loss of the fine-scale habitat mosaics created by traditional burning likely contributed to the extinction of small mammals in the region (Burrows *et al.*, 2006). Many desert reptiles are habitat specialists, so large-scale wildfires may benefit some species whilst negatively affecting others (Pianka, 1986). Woinarski *et al.* (Chapter 25, this volume) give some examples of reptiles that are affected by inappropriate fire regimes. More research is needed to understand the interactions among fires, rainfall, grazing and predators in Australian deserts (Pianka & Goodyear, 2012). Managing fire regimes in isolated, uninhabited regions of arid Australia is logistically difficult, and will require communication and collaboration between pastoralists, national parks and Aboriginal landholders (Woinarski *et al.*, Chapter 25, this volume).

17.2.6 Changes to fire regimes in temperate regions

In temperate regions of Australia, there is little consensus about whether Aboriginal peoples used fire as a tool to manage vegetation, and whether fire regimes and vegetation have changed substantially since European colonisation (Pringle *et al.*, 2009). Nonetheless, analysis of charcoal deposits and the records of early settlers suggest that

Aboriginal peoples used fire frequently in some parts of southern Australia (Black *et al.*, 2008; McLoughlin, 1998). By contrast, Europeans adopted a policy of fire suppression to protect property and grazing lands; such policies resulted in changes in the severity and extent of wildfires (Shea *et al.*, 1981). Whether these changes have affected reptiles remains unknown, although some species could be disadvantaged by fire-mediated vegetation changes (Pringle *et al.*, 2009, 2012). Recent studies suggest that, unlike mammals, reptiles do not show predictable responses to fire (Lindenmayer *et al.*, 2008); hence, maintaining a mosaic of habitats with different fire histories may be the best strategy for conserving reptiles (Driscoll & Henderson, 2008). Implementing effective fire management in temperate Australia remains a major challenge for reptile conservation.

17.2.7 Livestock grazing

Livestock grazing on natural vegetation occurs on 4.2 million square kilometres, or 55% of Australia. Grazing is the dominant land use on semi-arid and arid regions of the country, which are significant hotspots for reptile diversity (Pianka, 1969; Morton & James, 1988). Livestock grazing has resulted in loss of native vegetation, soil erosion, the degradation of riparian areas, and has contributed to declines of small mammals and birds (Martin & McIntye, 2007; Legge *et al.*, 2011). Although the impacts of grazing on reptiles are less clear, heavy grazing can cause reductions in reptile abundance at small spatial scales (James, 2003). For example, the heavily grazed and trampled bare ground (the piosphere) which surrounds bore-fed watering points is unsuitable for reptiles that rely on shrub layers or litter for cover, and may increase the risk of predation by aerial predators (James *et al.*, 1999). The provision of artificial watering points for cattle in arid Australia has also facilitated the spread of invasive cats, foxes (James *et al.*, 1999) and cane toads (Florance *et al.*, 2011), which can negatively affect reptile populations. Despite the ecological problems associated with livestock grazing, rangelands nonetheless contribute substantially to reptile conservation at regional scales (Woinarski *et al.*, 2013).

17.2.8 Predation by feral cats

Cats were deliberately released in Australia to control mice and rabbits in the nineteenth century, and they have since spread across the entire continent. Predation by feral cats is listed as a key threatening process under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act). Feral cats are a potential threat to vulnerable or threatened vertebrates, including at least 35 species of birds, 36 mammals, seven reptiles and three amphibians. Threatened reptiles at risk from cat predation include three Christmas Island lizards (the blue-tailed skink *Cryptoblepharus egeriae*, the forest skink *Emoia nativitatis* and Lister's gecko *Lepidodactylus listeria*), one legless lizard (striped legless lizard *Delma impar*), four skinks (great desert skink *Egernia kintorei*, Arnhem Land skink *Egernia obiri*, the Blue Mountains water skink *Eulamprus leuraensis*, and the Corangamite water skink *Eulamprus tympanum marnieae*) and the broad-headed snake *Hoplocephalus bungaroides* (Smith *et al.*, 2012, DEWHA, 2008a). Although cats have broad diets, and preferentially consume small mammals and birds, they can be significant predators of reptiles (Paltridge *et al.*, 1997). The recent decline of small mammals from savanna landscapes in northern Australia may result in cats including more reptiles in their diets. Potentially, cat predation could cause local extinctions of species which have suffered population declines due to cane toad poisoning, changes in fire regimes (Woinarski *et al.*, 2011), or other threatening processes. Species likely to be particularly vulnerable to cat predation in this respect are bluetongue lizards, which have suffered precipitous declines across northern Australia (Price-Rees *et al.*, 2010).

Controlling feral cats on the mainland is extraordinarily difficult due to their reluctance to consume toxic baits, and their low population densities in many landscapes. By contrast, feral cats have been eradicated on sub-Antarctic Macquarie Island and on the Montebello Islands off Western Australia (Nogales *et al.*, 2004). On the mainland, the best way to minimise the impacts of feral cats on vulnerable reptiles is to maintain appropriate fire regimes and habitats (i.e. cover) in the landscape; the possibility that dingoes might suppress cat (and fox) densities also warrants further investigation (Dickman, 1996).

17.2.9 Predation by European red foxes

The red fox was deliberately introduced to Australia in 1855, and has since spread across much of Australia (Dickman, 1996). Foxes prey on a diversity of animals, but are a major predator of small- and medium-sized mammals, ground-nesting birds and chelid turtles (Dickman, 1996). Predation by the European red fox is listed as a key threatening process under the EPBC Act. Foxes pose a threat to at least 12 species of threatened reptiles, including four species of marine turtle (loggerhead turtle *Caretta caretta*, green turtle *Chelonia mydas* (Figure 17.4, Plate 36), leatherback turtle *Dermochelys coriacea*, flatback turtle *Natator depressus*), four species of freshwater turtle (western swamp turtle *Pseudemydura umbrina*, Fitzroy River turtle *Rheodytes leukops*, Mary River turtle *Elusor macrurus* and Bellinger River turtle *Emydura signata*), two skinks (corangamite water skink *Eulamprus tympanum marnieae* and the great desert skink *Egernia kintorei*), one legless lizard (striped legless lizard *Delma impar*) and the broad-headed snake *Hoplocephalus bungaroides* (DEWHA, 2008b).

Reducing the impacts of foxes on native wildlife can be achieved with the use of extensive fencing or broad-scale 1080 poison baiting (Short & Turner, 2000). Baiting has been very successful in Western Australia, but less successful in eastern Australia (Saunders *et al.*, 2010). Fox control is expensive, and must be monitored and continued indefinitely to be successful. Increasing landholder participation and use of more efficient baiting techniques, such as broad-scale aerial baiting, are necessary to improve fox control in eastern Australia (Saunders *et al.*, 2010).

17.2.10 Invasive fire ants and yellow crazy ants

Two species of highly invasive ants, the crazy ant *Anoplolepis gracilipes* and the fire ant *Solenopsis invicta* pose a potential threat to Australian reptiles. Crazy ants form large nests and super-colonies that can cover large areas (750 ha) and they vigorously attack animals that disturb nests (Abbott, 2006). Although they lack stings, crazy ants kill invertebrates and vertebrates by biting and spraying formic acid. The ants prey on arthropods, earthworms, molluscs, land crabs, birds, mammals and reptiles (O'Dowd *et al.*, 2003). In Australia, crazy ants inhabit Christmas Island and a 2500 km^2 region of Arnhem Land (Young *et al.*, 2001). The ants thrive in human-disturbed areas, and can inhabit tropical and subtropical habitats, grasslands, savanna woodlands, woodlands and rainforests (O'Dowd *et al.*, 2003). Climatic modelling suggests that this species could inhabit most of northern Australia, eastern Queensland and parts of northern New South Wales (Chen, 2008).

The impacts of crazy ants have been particularly severe on Christmas Island. The ants were introduced in the 1930s but remained in low numbers until the 1990s. However, by 2002, the ants had formed super-colonies, with densities of 2000 ants per m^2 covering 2500 ha of the islands' forests (Abbott, 2006). The forest floor was literally crawling with carnivorous crazy ants, and was not the sort of place you would want to picnic in. The prolific crazy ants soon decimated the red crabs (*Gecarcoidea natalis*), the dominant consumers of the forest floor, which led to massive changes within rainforest habitats (O'Dowd *et al.*, 2003). Crazy ants threaten endemic Christmas Island reptiles, which have undergone massive population declines in recent decades (Box 17.4). To control crazy ants, Parks Australia carried out aerial baiting of super-colonies in 2002, 2009 and 2012. The baiting programme in 2009 was highly successful, with ant densities reduced by 99% at super-colony sites (Boland *et al.*, 2011). Nonetheless, continued monitoring and baiting will be necessary to control the ants on Christmas Island.

Box 17.4 The trouble with islands

Although no Australian reptile species has 'officially' gone extinct since European settlement, many are perilously close and several are almost definitely 'unofficially' extinct. The Australian reptile fauna has seemingly done better than other continents, with some authors suggesting that 19% of all reptile species worldwide are now vulnerable to extinction (Bohm *et al.*, 2013). In Australia, 58 reptile species are currently listed as Threatened (see Table 17.1), or about 6% of the approximately 970 current species (EPBC, 2013).

Island species are particularly vulnerable to extinction, with all 22 confirmed reptile extinctions worldwide occurring on islands (IUCN, 2013). The first Australian reptile extinctions will probably also be on islands. The Australian Indian Ocean territory of Christmas Island has seen a rapid and catastrophic decline in five of its six native reptile species (Smith *et al.*, 2012). Over 60% of the island (85 km^2) is a National Park, but the ecological changes wrought by super-colonies of yellow crazy ants (*Anoplolepis gracilipes*) together with introduced plant and animal species are immense (O'Dowd *et al.*, 2003). Hypotheses suggested for the rapid decline of the three skink, one gecko and a blind snake species from Christmas Island include habitat change, an introduced disease or pathogen, climate change, competition with introduced reptile species and exotic predators (Smith *et al.*, 2012; Maple *et al.*, 2012). Disease now seems unlikely (Hall *et al.*, 2011), but introduced predators like the giant centipede (*Scolopendra subspinipes*) and the specialist lizard-eating Asian wolf snake (*Lycodon capucinus*, introduced around 1987) are prime contenders. Four of the declining reptile species contracted to the south-western tip of the island, which is the most distant point from the single port facility and airport where the wolf snake presumably first arrived (Smith *et al.*, 2012). One needs only to consider the accidental introduction of the brown tree snake (*Boiga irregularis*) to Guam, which caused the extirpation of 13 of Guam's 22 native bird species, to realise how much we underestimate the ability of snakes to locate and devour naïve prey (Rodda & Savidge, 2007).

On Christmas Island the two *Emoia* species and the blue-tailed skink have only recently gone: the coastal skink (*E. atrocostata*) was last seen in 2004, the forest skink (*E. nativitatis*, Figure 17.3, Plate 35) and the blue-tailed skink (*Cryptoblepharus egeriae*) in mid 2010 (Smith *et al.*, 2012). Despite intensive surveying, tiny Lister's

Box 17.4 (continued)

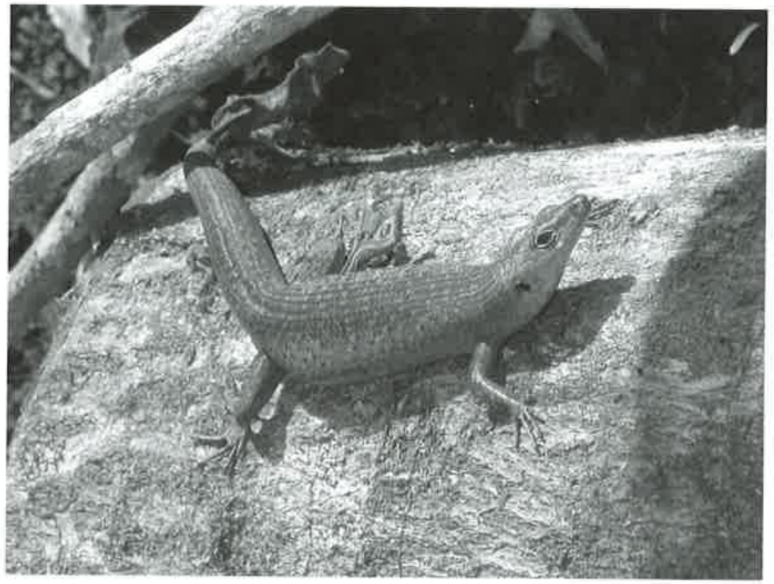

Figure 17.3 (Plate 35) This captive Christmas Island forest skink (*Emoia nativitatis*) may be the last individual, as despite extensive field surveys this species has not been seen in the wild since mid 2010. Photograph by Peter Harlow. A black and white version of this figure will appear in some formats. For the colour version, please refer to the plate section.

gecko (*Lepidodactylus listeri*) has not been located in the wild since late 2012. Christmas Island National Park staff were helplessly watching and monitoring these declines, and in 2009–10 they captured 64 blue-tailed skinks and 43 Lister's geckos to begin captive breeding colonies. By late 2014 over 400 blue-tailed skinks and 200 Lister's gecko were in captivity on Christmas Island and at Taronga Zoo, Sydney. But what should be done with these lizards, as the threatening processes have not been identified, and thus no safe habitat remains to release them on Christmas Island? One solution may be 'assisted colonisation', to release these captive populations on another small tropical island.

The remarkable Pedra Branca skink (*Niveoscincus palfreymani*) survives in about 0.14 ha of available habitat on a 2.5 ha rock 26 km off the south-east coast of Tasmania. Three population estimates over 14 years show the population varies from about 290 to 560 individuals (Brothers *et al.*, 2003), and most evidence suggests that this island has been separated from Tasmania for at least 19 000 years (Banks, 1993). This species is a contender for natural extinction, with no human assistance, in the next few millennia as each stochastic population decline and genetic bottleneck increases the likelihood of inbreeding depression. One study that investigated the frequency and severity of catastrophic die-offs in 88 species of vertebrates suggested that the probability of a 50% or greater population decrease in any one year is approximately 14% per generation, or about one in every seven generations (Reed *et al.*, 2003). Biologists today are faced with increasing political and philosophical decisions; should we translocate Christmas Islands' captive lizards, or perhaps Pedra Branca skinks, to new islands or let extinction occur?

The invasive fire ant *Solenopsis invicta* builds earthen mounds that harbour between 200 000 and 400 000 workers, and can attain densities of up to 2600 mounds per hectare. The ants possess a powerful venom, and when attacking en masse, they can kill vertebrates, stock, domestic pets and humans (Moloney & Vanderwoude, 2003). In Australia, fire ants cover approximately 50 000 ha of the south-western suburbs of Brisbane and the eastern suburbs of Ipswich in south-eastern Queensland (Schmidt *et al.*, 2010). Already, the ants have reduced the abundance of invertebrates and reptiles, and are poised to invade the coastal belt and the more mesic inland areas of Australia. The spread of fire ants poses a significant risk to reptiles, particularly hatchling sea turtles and ground-dwelling lizards (Moloney & Vanderwoude, 2003).

To reduce the spread of fire ants, the Queensland Government has implemented movement controls to individuals and commercial operators in areas containing fire ants. The Australian Government has funded aerial detection of nests and deployment of baits in an attempt to eradicate fire ants from the Brisbane region. In Yarwun, central Queensland, fire ants were successfully eradicated. Nonetheless, continued surveillance and eradication programmes are necessary to prevent the spread of fire ants (Schmidt *et al.*, 2010).

17.2.11 Climate change and sea turtles

Climate change poses a major threat to Australian reptiles, particularly species which exhibit temperature-dependent sex-determination or which depend on rainfall for survival. For example, changing rainfall patterns in Western Australia threaten the survival of the critically endangered western swamp turtle, *Pseudemydura umbrina* (Box 17.5). Sea turtles in Australia could be especially vulnerable to climate change because many populations are depleted, or are harvested by traditional hunters (Box 17.6). Further population perturbations could push some populations to extinction, and thus understanding how and why sea turtle populations are vulnerable to increasing temperatures associated with climate change is an urgent conservation problem (Hamann *et al.*, 2013). Like all ectotherms, temperature influences every facet of the life history and ecology of sea turtles. This includes embryonic survival, hatchling sex, hatchling body size and performance, determining the rates of physiological processes, and influencing foraging distributions, food availability, nesting distributions, and nest-site availability (Hamann *et al.*, 2013). Although sea turtles spend the vast majority of their lives in the ocean, the terrestrial

Box 17.5 The critically endangered western swamp turtle *Pseudemydura umbrina*: promising initial recovery following decades of slow decline

Western swamp turtles are restricted to two ephemeral swamps of marginal quality on the fringe of Australia's fastest growing city, Perth, Western Australia. Swamps usually fill and remain wet during winter, when the carnivorous turtles are aquatic and forage for prey, but begin to dry from late winter through summer. As swamps dry, turtles migrate to nearby terrestrial aestivation sites, usually comprised of natural tunnels underground or beneath surface debris (Burbidge & Kuchling, 2004; Burbidge, 1981). Swamp filling and drying cycles are strongly tied to seasonal rainfall, which has

Box 17.5 (continued)

declined over the past three decades (Burbidge & Kuchling, 2004; Mitchell *et al.*, 2012a). In many recent years the ponds have dried before females are able to accumulate sufficient energy stores to produce eggs, resulting in the absence of population-level reproduction in those years (Mitchell *et al.*, 2012a,b).

Western swamp turtles have the slowest life history of any Australian turtle, which combined with a current population size of <50 adults in the wild, renders it vulnerable to extinction (Burbidge, 1981; Mitchell *et al.*, 2012a). Females are smaller than males, and reach maturity at 11–15 years of age and can live in excess of 60 years. During reproductive years, females lay a single clutch of only three to five eggs, but reproduction is strongly linked to environmental conditions and is thus less than annual (Burbidge & Kuchling, 2004; Burbidge, 1981). Variable and unpredictable seasonal rainfall contributes to slow growth rates, leading to delayed maturity and irregular and stochastic reproduction (Mitchell *et al.*, 2012b; Burbidge & Kuchling, 2004; Burbidge, 1981).

Western swamp turtles recently were on a trajectory towards extinction. At the larger of the two known populations (Twin Swamps Nature Reserve), the number of adult turtles known to be alive decreased from 38 in 1963 to only seven by 1984, an average loss of just over one adult per year (Burbidge & Kuchling, 2004). Clearly, this population decline and the current low number of adult animals exemplify that the survival and reproduction of every individual turtle is crucial to maintaining the entire species. There is hope for this species in the wild, however, because captive assurance colonies are now supplementing wild populations. Although the Twin Swamps population stayed below 10 individuals through 2001, this population has increased rapidly because of conservation efforts (Burbidge & Kuchling, 2004).

The rapid recovery of this species is encouraging, but increasing aridity could hinder population growth rates by constraining reproduction and foraging opportunities. Annually, turtles spend six or more months aestivating in terrestrial environments. The migration to and from wetlands is a period of high predation (Burbidge & Kuchling, 2004), and terrestrial aestivation substantially increases vulnerability to desiccation, energy depletion, and hyperthermia (Burbidge, 1981; King *et al.*, 1998). Two novel approaches are being used to: (1) predict how increasing temperatures and shorter, more variable hydroperiods could impact individual turtle growth rates (which influences age at maturity); and (2) identify wetland sites that will maintain favourable hydroperiods under climate change, and potentially translocate turtles to these sites. Increased water temperatures could increase growth rates of the hatchling and juvenile life stages, potentially allowing individuals to reach maturity at earlier ages (Mitchell *et al.*, 2012b). Assisted colonisation to high-quality ephemeral wetlands could help establish long-term, viable populations that are robust to the impacts of climate change. Selecting appropriate release sites can be difficult, but several candidate wetlands seem promising for establishing new populations (Mitchell *et al.*, 2012a). Anticipating the effects of climate change, and preparing for them will help ensure that western swamp turtle populations remain in the wild far into the future.

Box 17.6 Conserving endangered sea turtles and cultural values: the complexities of contemporary harvest

One challenging – and very real – goal is to balance science-based protection efforts with traditional use of wildlife by indigenous peoples (Kwan *et al.*, 2001; Wilson *et al.*, 2010; Nursey-Bray, 2009; Butler *et al.*, 2012). In Australia, the *Native Title Act* defines the rights of Indigenous Australians to continue traditional practices, even when these practices may be prohibited by contemporary law. Concerns over declining sea turtle populations led to a closure of turtle and egg harvest in Queensland in July 1968 through enactment of the Queensland Fisheries Act (Miller & Limpus, 2012). The *Native Title Act*, passed in 1993, reinstated the rights of Indigenous Australians to use native animals (including sea turtles and eggs, dugong and other endangered species) legally for communal, non-commercial purposes (Butler *et al.*, 2012; Kwan *et al.*, 2001). Traditional hunting provides an important, and under-utilised, opportunity for ecologists to learn from communities with extensive knowledge of ecology and animal behaviour.

Sea turtles and their eggs are culturally important foods for Torres Straight Islanders (Butler *et al.*, 2012). Green turtles *Chelonia mydas* are preferred (Figure 17.4, Plate 36), but other species and their eggs are also consumed (e.g. hawksbill *Eretmochelys imbricata*, flatback *Natator depressus*; olive ridley, *Lepidochelys olivacea*; (Butler *et al.*, 2012)). Female sea turtles are targeted because of their high fat content (Kwan *et al.*, 2001); thus, hunters selectively remove individuals that could have otherwise continued to lay eggs for decades. Information on the level of harvest is lacking, and we do not yet understand the potential impacts of traditional hunting on severely

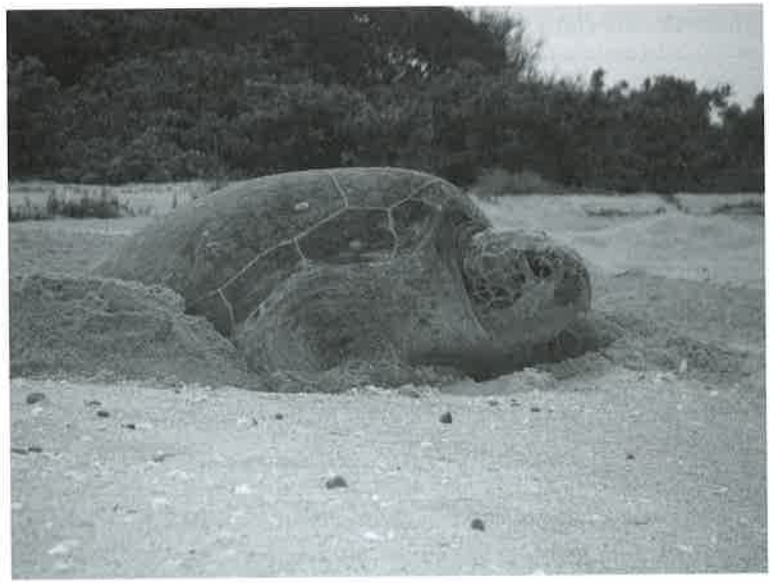

Figure 17.4 (Plate 36) The green sea turtle (*Chelonia mydas*) is the most widespread of the sea turtles nesting in Australia, and is thus exposed to a wide range of climatic conditions and local threats, including hunting by indigenous groups. Although the impacts of climate change and traditional hunting on sea turtles is generally unknown, continuing efforts to protect this species will require an integrative approach that can minimise multiple threats. Photograph by Ian Bell. A black and white version of this figure will appear in some formats. For the colour version, please refer to the plate section.

Box 17.6 (continued)

reduced populations (Miller & Limpus, 2012; Kwan *et al.*, 2001). One concern raised by opponents of indigenous harvest is the manner by which turtles are located, captured, and despatched; today new technologies (motorboats, metal harpoons, knives for butchering) substantially increase harvest success. In some instances, however, animals are not despatched immediately, which has led to strong debate in the media about the ethical nature of traditional harvest. In 2012, Queensland closed loopholes to ensure that traditional harvest complies with animal cruelty laws. Although this debate is far from over, the larger conservation issue is whether sea turtle populations can withstand the pressures of traditional harvest when combined with climate change and other threats (Hamann *et al.*, 2013).

What are current, feasible solutions to ensure that sea turtle populations can still recover, while maintaining indigenous harvest? The latest approaches to responsible and sustainable management focus on blending scientific and traditional indigenous knowledge (Wilson *et al.*, 2010). This can enable local communities to develop their own harvest management plans using scientific input (e.g. self-imposed seasonal closures, restricted areas, catch limits, restrictions on harvest methods (Wilson *et al.*, 2010; Kwan *et al.*, 2001)), combined with community monitoring of the impacts to populations. Assisting traditional groups to develop their own adaptive management plans fully – which includes self-monitoring and applying traditional penalties for breaches – with the aid of scientific input provides enormous opportunities for conservation of species, ecosystems and culture. These benefits could not be achieved any other way, and provide a promising glimpse into a future where contemporary legislation and policy incorporate indigenous knowledge and values to maintain and conserve biodiversity.

environment is crucial for reproduction. Sea turtles bury their eggs on sandy beaches in tropical to temperate regions, and the incubation temperature of the nest influences hatching success and sex. Climate change is predicted to increase ambient temperatures above that of current decades by 1–5 °C by 2070 in Australia (Cabrelli *et al.* Chapter 4, this volume), which has the potential to influence sea turtle nest temperatures and population-level primary sex ratios. A global sea-level rise of up to 79 cm by 2100 could threaten several important nesting populations by reducing the availability of nesting habitat on low-lying islands or in areas limited by human development. Increased precipitation or changes in the severity or intensity of tropical cyclones could also impact nesting beaches (Hamann *et al.*, 2013). Changes in water temperatures could also influence the distribution and availability of food, migratory paths, inter-nesting intervals, and individual growth rates (Hamann *et al.*, 2013). Although some work has made progress in understanding these threats overseas (Witt *et al.*, 2010), Australian studies have yet to tackle climate change impacts at foraging grounds or on the potential changes in migratory routes. Sea turtles provide a wide range of ecosystem services, and protecting these charismatic megafauna under climate change will allow the continuation of important ecological, social, cultural and economic services, not only in Australia, but worldwide.

17.3 Conclusions

17.3.1 The challenges of conserving reptiles in a vast continent

Australia harbours a rich and diverse reptile fauna, which presents substantial challenges to conservation. Australia covers an area of 7 688 503 square km, and the dominant land use is livestock grazing on natural vegetation, which occurs on 55% of the continent. By contrast, only 7% of the continent is devoted to conservation reserves, while other protected areas, including indigenous uses, cover 13% of Australia. The current reserve system does not adequately protect threatened reptiles, nor is it likely to do so substantially in the future (Watson *et al.*, 2011). Hence, to conserve Australian reptiles, we need a coordinated approach which transcends land tenure and State/Territory boundaries (Woinarski & Fisher, 2003). This complex task requires setting clear long-term goals, installing appropriate monitoring programmes, and engaging in adaptive experimental management. Managers will need to respond to dynamic changes within systems, and account for future changes that are likely to occur under climate change (Lindenmayer & Hunter, 2010). Ultimately, reptile conservation will require goodwill and effective communication between a diversity of stakeholders; the WALFA and Ecofire projects demonstrate that this is possible.

Reptile conservation in Australia will increasingly be in the hands of the private sector, non-government organisations and concerned citizen groups. Many State and Territory wildlife departments have been endlessly 'restructured', and funding which could have been directed to staff salaries or conservation efforts has been funnelled to needless name changes on websites, logos and stationery. Indeed, many parks and wildlife departments have been so starved of funding that they can no longer adequately manage their own National Parks. The increasing activity of fringe animal rights groups has seen many State and Territory Wildlife Departments begin inane and resource-wasting bureaucracies that prescribe and legally enforce cage size regulations for pet lizard keepers! One has only to consider the thousands of reptiles that die each year on our roads, or from vegetation removal, to see the idiocy of such regulations.

The paucity of information concerning the effects of fire, grazing and introduced predators on reptiles, coupled with the absence of natural history data for many species, creates additional problems for conserving reptiles (Cogger *et al.*, 1993). Many threatening processes vary across broad biogeographic regions, so conservation actions must often be tailored to specific localities, and to particular species (see Boxes 17.1, 17.2 and 17.3). For example, Parks Australia's captive breeding programmes in partnership with Taronga Zoo have been implemented to prevent the extinction of imperilled Christmas Island reptiles (see Box 17.4). Captive breeding may also be a necessary step to prevent the extinction of the other species, such as the grassland earless dragon (*Tympanocryptis pinguicolla*), which has declined precipitously in recent years (Dimond *et al.*, 2012). More research is necessary to determine the causes of these recent declines. Despite the recent claim that conservation biologists do not need to collect any more data (Possingham, 2012), we clearly need more detailed natural history studies on Australian reptiles. Without basic information on the habitat requirements, diets, life history, and patterns of dispersal of threatened reptile species, it is difficult to diagnose, let alone reverse, population declines (Caughley & Gunn, 1996). Finally, we need to engender an awareness and appreciation of Australia's unique reptile fauna among all young Australians, who will ultimately be responsible for conserving our future.

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