



Rapid increase of Australian tropical savanna reptile abundance following exclusion of feral cats



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ABSTRACT

Feral cats have been responsible, in part, for the extinction of many species of mammal, bird and reptile globally, especially on islands. Whilst there is extensive evidence of the predatory impacts of cats on mammals and birds, far less is known about their ecological impacts on reptiles, especially in continental situations. We conducted a field experiment to evaluate the impact of feral cats on terrestrial vertebrates in tropical savanna environments of northern Australia. Three experimental treatments were applied to six 64 ha plots to compare and contrast responses of reptile abundance and species richness to predator exclusion and the additive effects of frequent fire. Replicated pitfall-trapping was undertaken in each plot on seven sampling occasions between November 2013 and November 2015. We analysed relative abundance and species richness data using generalized linear mixed models. There was a significant increase in the abundance of reptiles over a two year period in cat-excluded plots with reptile abundance increasing at twice the rate in cat-exclusion plots compared with cat-accessible plots and there was an additive effect of time-since-fire. Cat exclusion had a positive effect on reptile species richness over time, however the evidence for this pattern was weak when seasonal variation was taken into account. Predation by cats, in synergy with other disturbance processes, could adversely impact reptile species and communities elsewhere in the world where feral cats have been established and warrants further investigation.

1. Introduction

Invasive mammalian predators are significant drivers of biodiversity loss worldwide (Doherty et al., 2016). In particular, feral cats have caused high rates of mortality for many bird and mammal species (Loss et al., 2013), and for the extinction of at least 18 species of island-endemic vertebrates (Medina et al., 2011; Nogales et al., 2013). Within Australia, predation by feral cats has been identified as a major contributing factor in the marked and widespread declines of northern Australia's mammal fauna in recent decades (Fisher et al., 2014; Woinarski et al., 2011).

Feral cats now occupy the entire Australian mainland and numerous offshore islands (Legge et al., 2016). Their population densities vary considerably across the continent (Legge et al., 2016); however, even at low densities, cats can deplete local populations of small and medium-sized mammals (Vázquez-Domínguez et al., 2004), increasing the risk of extinction to threatened species (Moseby et al., 2015).

Whilst extensive evidence of the predatory impacts of cats on mammals and birds has accumulated (Loss et al., 2013; Medina et al., 2011; Woinarski et al., 2017), far less is known about their ecological impacts on ectotherms. Reptiles and amphibians form a large component of the diet of cats globally (Bonnaud et al., 2010; Medina et al., 2011; Woods et al., 2003) and within Australia (Kutt, 2012; Read and Bowen, 2001; Woinarski et al., 2018). Several island endemic reptile species have become threatened through predation by introduced cats (Arnaud et al., 1993) and competition for food resources (Donlan et al., 2000). Dietary studies indicate that feral cats prey on a wide taxonomic range of Australian mainland reptile species and are capable of switching prey as preferred species are depleted (Dickman and Newsome, 2015; Doherty et al., 2015). However, the broader ecological impacts of cat predation on reptile populations and communities are uncertain.

To assess the ecological effects of feral cat predation, it is necessary to quantify their predatory impact on natural populations and species

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assemblages, where possible in the context of other environmental influences. An understanding of the impact of predation by feral cats relative to, or in synergy with, other potential drivers of biodiversity loss is desirable in order to make informed decisions about where and how to best direct management interventions. Studies in northern Australia have shown that cats preferentially hunt in landscapes affected by recent wildfire and grazing (McGregor et al., 2017; McGregor et al., 2016) where they have greater hunting success on mammalian prey (Leahy et al., 2016; McGregor et al., 2015). These findings suggest important interactions between feral cat predation and altered fire regimes driving declines of northern Australian small and medium sized mammals. Many Australian reptile species and communities are also sensitive to altered fire regimes (e.g. Hu et al., 2013; Legge et al., 2008; Pianka and Goodyear, 2012; Trainor and Woinarski, 1994; Valentine and Schwarzkopf, 2009). The interactive effects between feral cat predation and altered fire regimes may also impact reptile diversity.

We conducted a field experiment to evaluate the predatory impact of feral cats on northern Australian savanna terrestrial vertebrates. Here we report on the responses of reptiles to experimental exclusion of feral cats. We compared population responses of reptiles in replicated fenced plots that excluded cats to unfenced plots accessible to cats. As fire is a ubiquitous and important component of Australian tropical savanna woodland ecosystems, we attempted to manage fire as part of the experimental design. We predicted, firstly, that if feral cats exert significant predatory pressure on reptile communities, then populations should increase when cats are excluded. Secondly, we predicted that if frequent fire facilitates increased feral cat predation of reptiles, then when exposed to feral cats, populations of reptiles should increase with reduced fire frequency compared to those with frequent fire.

2. Methods

2.1. Study area

The study was conducted in Kakadu National Park (E 132°22.47, N 12°38.97) – the largest terrestrial national park in Australia and a World-Heritage Area (Fig. 1). The climate is monsoonal and is characterised by a humid wet season between December and March, during which the majority of the ca. 1500 mm mean annual rain falls. We undertook the study at Kapalga, north-western Kakadu, which consists of lowland open forest dominated by *Eucalyptus miniata* and *E. tetradonta* on flat topography. The tropical savanna of Kakadu National Park is frequently burnt with fires typically > 1 km² in extent and return intervals of less than three years: i.e. at least 33% (and often at least 50%) of the lowland forests are burnt every year (Russell-Smith et al., 2017). An extensive landscape-scale fire experiment was undertaken at Kapalga from 1989 to 1995 within large experimental compartments (Andersen et al., 2005). The area has also been the subject of extensive wildlife ecological studies and monitoring (e.g. Braithwaite and Muller, 1997; Griffiths et al., 2015; Woinarski et al., 2001). Consequently the faunal assemblage composition and fire history of the area are well documented.

2.2. Experimental design

Six 64 ha square plots with similar habitat and landscape characteristics and comparable fire histories over the previous 10 years (2002–2012, MODIS fire scars, 250 m × 250 m resolution, North Australia and Rangelands Fire Information (NAFI), <http://www.firenorth.org.au>) were selected using topographic maps, satellite imagery, and vegetation mapping (Schodde et al., 1986). Three experimental treatments were applied to the plots to compare and contrast responses of reptile abundance and richness to predator exclusion and the additive effects of frequent fire, as follows:

1. Predator exclusion and fire suppression

2. No predator exclusion and fire suppression, and
3. No predator exclusion and no fire suppression.

A fully balanced design that incorporated a further treatment with ‘predator exclusion and no fire suppression was considered but not included, because of: (i) ethical concerns about lack of escape routes from fire for some animals inside enclosures; (ii) a high likelihood that fire in predator proof enclosures would damage fences directly and indirectly by causing treefalls; and (iii) a lack of availability of suitable locations within the study area for additional fenced sites with similar fire history and vegetation characteristics.

For treatment 1, predator exclusion fences with firebreaks 8 m in width were established around the perimeters of two plots. The fence design followed that successfully used at Arid Recovery in South Australia to exclude feral cats, foxes and rabbits (Moseby and Read, 2006). The fences were constructed using 50 mm hexagonal wire mesh, to a height of 1800 mm with a curved floppy section extending 450 mm outward from the top of the fence. Internal and external foot aprons extended 550 mm from the base to prevent animals from digging under the fence. Note that this fencing excluded not only cats, but also other large mammals present in the area (including dog *Canis familiaris*, Asian buffalo *Bubalus bubalis*, horse *Equus caballus*, pig *Sus scrofa* and agile wallaby *Notamacropus agilis*), and may have prevented large goannas *Varanus* spp. and large snakes from entering or exiting the fenced plots.

Treatments 2 and 3 were unfenced plots. Firebreaks 8 m wide were established around the perimeters of two plots for treatment 2; the remaining two plots had no fire breaks for treatment 3.

Fire suppression activities were undertaken by Kakadu Park management staff at all four plots with fire breaks, including early dry season fuel reduction burning around most of the external perimeters, and active fire suppression when necessary inside the plots. However, despite these measures, strict fire exclusion in line with the planned design was not achieved. All plots were burnt at least once between the months of May and November over the duration of the experiment (see Table 1).

2.3. Data collection

Initial (baseline) surveys for reptiles in each plot occurred in November 2013, prior to fence completion. After completion of fences in December 2013, each plot was surveyed three times annually in the late wet (March/April), mid-dry (June/July) and early wet (Oct/Nov) seasons until November 2015. Within each plot six transects, 200 m apart and 800 m in length, were established. Four 20 L pitfall trap buckets (290 mm diameter and 400 mm deep, drift fence 10 m long and 0.3 m high) were installed 200 m apart along each transect. Each transect was sampled for a 72 h period during each sampling session. Half of the transects (alternate transects) in each of three plots, one from each treatment, were sampled concurrently, followed by the second half of the transects over a total period of seven days. This procedure was then repeated for the other three plots, i.e. there was a total sampling effort of 72 pitfall-days per plot per sample event. The order of plots and transects sampled was varied between survey sessions. Traps were checked twice daily (c. 0700 h and 1700 h) and the species and number of individuals captured were recorded: note that trapped individuals were not marked, so the abundance tally may include recaptures. Unequal trapping effort occurred between some sampling periods due to heavy rain filling buckets and/or meat ant invasions of some buckets. Total trapping effort was > 90% across all plots except in three sampling sessions. In November 2013, 85% effort was achieved for three of the unfenced sites and 50% effort for one unfenced plot and both fenced plots. In March 2014, sampling effort in the fenced plots was 80 and 88%, and unfenced plots ranged from 83 to 85%. Lastly, in March 2015 two unfenced plots achieved 85% and 88% effort whilst all others were > 90%.

Both fenced plots, and three unfenced plots, were burnt prior to the

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