Journal for Nature Conservation 22 (2014) 577–585

Contents lists available at ScienceDirect

Journal for Nature Conservation

journal homepage: www.elsevier.de/jnc

Reptile bycatch in a pest-exclusion fence established for wildlife reintroductions

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ARTICLE INFO

Article history:
Received 2 June 2014
Received in revised form 27 August 2014
Accepted 27 August 2014

Keywords:
Predator-proof fence
Habitat fragmentation
Hotspots
Hot moments
Mortality
Dispersal

ABSTRACT

Conservation fences have been used as a tool to stop threatening processes from acting against endangered wildlife, yet little is known of the impacts of fences on non-target native species. In this study, we intensively monitored a pest-exclusion fence for 16 months to assess impacts on a reptile community in south-eastern Australia. We registered 1052 reptile records of six species along the fence. Encounters and mortality were greatest for eastern long-necked turtles (*Chelodina longicollis*), whereas impacts on lizards (*Tiliqua rugosa*, *Tiliqua scincoides*, *Pogona barbata*, *Egernia cunninghami*) and snakes (*Pseudonaja textilis*) were more moderate. We recorded several *Chelodina longicollis* recaptures at the fence and many of these were later found dead at the fence, indicating persistent attempts to navigate past the fence. We conservatively estimate that the fence resulted in the death of 3.3% and disrupted movements of 20.9% of the turtle population within the enclosure. Movement disruption and high mortality were also observed for turtles attempting to enter the nature reserve, effectively isolating the reserve population from others in the wider landscape. Of 98 turtle mortalities, the most common cause of death was overheating, followed by predation, vehicular collision, and entanglement. Turtle interactions were clustered in areas with more wetlands and less urban development, and temporally correlated with high rainfall and solar radiation, and low temperature. Thus, managers could focus at times and locations to mitigate impacts on turtles. We believe the impact of fences on non-target species is a widespread and unrecognized threat, and suggest that future and on-going conservation fencing projects consider risks to non-target native species, and where possible, apply mitigation strategies that maintain natural movement corridors and minimize mortality risk.

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Introduction

Conservation fences have been used worldwide as a tool to stop or ameliorate processes that threaten biodiversity (Dickman 2012; Hayward & Kerley 2009). Such fences are used to alleviate human–animal conflict, to reduce human persecution on threatened species, and to minimize the impact of introduced species (Hayward & Kerley 2009). Fences can provide *in situ* protection of threatened species, facilitate the reintroduction of threatened species, and provide opportunity for education, ecotourism and research (Dickman 2012). Despite their worldwide use, there is a geographic bias in the use of fences for conservation, with many examples in Australia, New Zealand and southern Africa. The threats in Australasia are largely introduced predators, whereas in Africa they arise largely from human–animal conflict (Hayward & Kerley 2009). Conservation fences can be very effective in protecting and conserving endangered wildlife, with many cases of native species recovery (Dickman 2012; Hayward & Kerley 2009).

The use of fences for pest management in Australia has a long history, initially consisting of fences to protect croplands against the European rabbit (*Oryctolagus cuniculus*) and livestock from dingos (*Canis lupus dingo*) (Pickard 2007a; Saunders, Gentle, & Dickman 2010). More recently, there has been an increase in the use of pest-exclusion fences for conservation purposes in Australia (Bode & Wintle 2009; Long & Robley 2004) to protect vulnerable native fauna from invasive predators such as the European fox (*Vulpes vulpes*), domestic and feral cats (*Felis catus*), and feral pigs (Doupé et al. 2009; Hayward & Kerley 2009; Long & Robley 2004; Moseby & Read 2006).

Pest-exclusion fences have clear conservation benefits for populations of endangered animals by controlling the spread of diseases

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http://dx.doi.org/10.1016/j.jnc.2014.08.014
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from domestic to wild populations, excluding exotic predators or competitors, and reducing human–animal conflicts (Hayward & Kerley 2009). However, the fences themselves can negatively impact non-target native wildlife by disrupting natural movement and dispersal processes, increasing mortality via entanglement and exposure, and enforcing inbreeding and isolation (Bode & Wintle 2009; Flesch et al. 2010; Hayward & Kerley 2009; Long & Robley 2004). Fences are also costly to build and maintain, with an opportunity cost for other conservation and management priorities (Scofield, Cullen, & Wang 2011).

While there are some reports of impact of fence design and alignment on select native mammal and bird populations (Hayward & Kerley 2009; van der Ree 1999), information on the impact of fences on reptiles is limited. Reptile mortality has been observed in feral animal-exclusion fences in Australia (Kuchling 2000; Long & Robley 2004) and South Africa (Burger & Branch 1994), particularly turtles, but the magnitude of the impact of such fences and the circumstances that trigger encounters and mortalities remain unclear. The impact of such fences could be highest for vagile animals, as species requiring frequent movements are more likely to encounter fencing and become isolated from critical resources or exposed to mortality risk. One such species that is common in our study system is the eastern long-necked turtle (*Chelodina longicollis*), which travels overland to nest, estivate, and move between wetlands in response to wet-dry cycles–behaviors that are essential for survival of individuals and the elements of population dynamics that support their persistence (Rees, Roe, & Georges 2009; Roe, Brinton, & Georges 2009). Other mobile terrestrial species, such as large lizards and snakes, may also be disturbed by a fence and suffer high mortality, as they also move extensively through the landscape (Cogger 2000; Fergusson & Algar 1986; Price-Rees, Brown, & Shine 2012; Whitaker & Shine 2003).

Here, we evaluate how a pest-exclusion fence affects non-target wildlife at a site in south-eastern Australia. We assess the effect of fences on movements and mortality in a reptile community, and environmental factors that explain these parameters that may be used to predict times and locations of highest concern. Such information can guide land managers in mitigating the impact of fences on non-target native wildlife, and in better assessing the trade-off between costs and benefits of fence projects.

Methods

Study area

Our study site was in Mulligans Flat Nature Reserve, located in the Australian Capital Territory (ACT) of Australia. The 791 ha reserve is part of a large-scale woodland restoration project (Manning et al. 2011), around which an 11.5 km-long pest-exclusion fence was constructed in 2009. The fence design was based on similar fences in Australia (Moseby & Read 2006). The fence is electrified and stands 1.8 m high with seven plain wires supporting rabbit mesh (30 mm), with a 60 cm "floppy overhang" and netting buried to a width of 45 cm on either side (Fig. 1). Several self-closing gates are placed in the fence perimeter which allows visitors and park maintenance staff to pass (Shorthouse et al. 2012). The goal of the pest-exclusion fence is to protect native fauna and flora within the fenced boundaries, to facilitate re-introduction of locally extirpated species, including the Eastern Bettong (*Bettongia gaimardi*) and the Southern Brown Bandicoot (*Isoodon obesulus*), and to exclude the introduced fox, domestic cats and dogs, as well as hares and rabbits from the sanctuary (Manning et al. 2011; Shorthouse et al. 2012).

Fig. 1. Pest-exclusion fence at Mulligans Flat Nature Reserve, Australian Capital Territory, Australia (Photo Credit: Larissa Schneider).

Fence monitoring

We monitored the fence by slowly driving (15 km/h) along an adjacent service road, which is located 2 m away from the fence. Information on turtle encounters was collected from January 2012 to April 2013 and expanded to include lizards and snakes from March 2012 to April 2013. We monitored the fence twice per week during the season when reptiles are typical active (September–April), and once per month during the overwintering period (May–August).

Whenever a reptile was sighted by the fence, we identified the species and registered its location using a hand-held GPS unit (Garmin 43434) and recorded its position along the fence (inside/outside) and its status (dead, injured, alive). We recorded encounters up to 10 m away from each side of the fence. If the animal was dead, we recorded the likely cause from external evidence observed on the animal (damage, lesions) or on the basis of context (entrapped, overheated, crushed). All dead reptiles were removed from the fence.

We marked turtles with unique codes by notching the shell (Kennett & Georges 1990), and measured maximum carapace length (CL) and midline plastron length (PL) with calipers (±0.1 mm) and body mass with a scale (±5 g). Turtles with a CL < 145 mm were considered juveniles; those for which CL > 145 mm were classified as males or females on the basis of external morphological features (see Kennett & Georges 1990). We did not mark or measure lizard and snakes, as our intention for these groups was not to estimate the number of animals affected by the fence, but instead to record frequency of encounters to determine location and time-specific hotspots and hot moments. All live animals were released at their point of capture on the same side of the fence.

Pond sampling

To assess the magnitude of impact for the fence on the wider population, we surveyed turtles from a subset of ponds in the fence vicinity. We trapped turtles in five nature reserve ponds inside the fence and three ponds outside of the fence. In each pond, we set four traps baited with sardines and liver once per month (5 consecutive days of trapping per month) from January 2012 to March 2013. More details on trapping methods are discussed by Roe, Rees, and Georges (2011). Turtles in the ponds were marked and measured in the same way as along the fence.

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