



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



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### 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 dingoes (*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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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 (*Isodon 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 ( $\pm 0.1$  mm) and body mass with a scale ( $\pm 5$  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.

## Data analysis

Statistical analyses were performed with SPSS (Version 21) and SAS Version 9.1 (SAS Institute 1999). The assumptions of normality and homogeneity of variances were checked by analysis of residuals and when data failed to meet these assumptions, data were transformed to approximate normal distributions and equal variances. Statistical significance was accepted at the  $\alpha=0.05$  level unless specified otherwise.

To assess spatial distributions of encounters, we plotted encounter records on a map of the fence using the ArcGIS (version 9.3.1: ESRI 2009). We then divided the fence into 500 m sections, resulting in 23 fence segments. Records of wildlife encounters were then summed by species and fence position (inside/outside) within each segment. To examine if reptiles had a random or non-random distribution around the fence, we used chi-square tests with the null-hypotheses that abundance of records within each fence segment would be equal.

To assess spatial correlates of reptile encounters (i.e. hotspots) within each fence segment, we selected the segment midpoint and created a polygon with 700 m radius. This distance was based on typical movement distances of *C. longicollis* determined from previous studies in the region, which would also cover the range of the other reptile species with smaller home ranges (Fergusson & Algar 1986; Rees, Roe, & Georges 2009; Roe & Georges 2007; Roe, Brinton, & Georges 2009). We used a vegetation map developed by Rees, Roe, and Georges (2009), with corrections using a 2012 aerial photograph. We then measured the area coverage of five land cover types (farmland, grassland, woodland, wetland, and urban development) and calculated correlations with number of animals encountered inside plus outside within each polygon. We performed multiple stepwise linear regressions to assess if the land cover variables (predictor variables) were associated with number encountered (response variable). In the above analysis, we log (ln) transformed all abundance estimates.

To assess temporal distributions of encounters, we summed the species records monthly. To determine if temporal distribution of records were random or non-random, we used chi-square tests with the null-hypotheses that abundance of records within each month would be equal. To examine correlates of temporal distributions of encounters with environmental variables (i.e. hot moments), we considered reptile encounters (alive and dead) in relation to daily maximum temperature ( $^{\circ}\text{C}$ ), rainfall (mm) and daily solar exposure ( $\text{MJ}/\text{m}^{-2}$ ). We considered the number of reptile records in the fence within an interval of 15 days. We used the mean value for the 15-day period for daily maximum temperature and daily solar exposure, and for rainfall we used the amount accumulated over the period. We then performed multiple stepwise linear regressions to determine the best supported model of association between the environmental variables and the number of individuals encountered for each species. We log (ln) transformed turtle records and square root (sqrt) transformed lizard to better meet assumptions.

To assess if variables would influence mortality, we performed a logistic regression using dead turtle encounters as the response variable. As explanatory variables we considered location (inside or outside of the fence), season (summer 2012, autumn 2012, winter 2012, spring 2012, summer 2013, and autumn 2013), sex (male, female, juvenile), carapace length (cm), daily maximum temperature ( $^{\circ}\text{C}$ ), rainfall (mm), and daily solar exposure ( $\text{MJ}/\text{m}^{-2}$ ). We considered an interval of 15 days for each of these environmental variables as described above.

To examine whether fence bycatch risk depended upon demographic group, we compared size-frequency distributions and sex ratios of turtles encountered inside versus outside the fence, in addition to all turtles from the fence (inside plus outside) versus

**Table 1**

Live and dead reptile encounters in the pest-exclusion fence at Mulligans Flat Nature Reserve, Australian Capital Territory, Australia.

	Records along the fence			Deaths along the fence		
	Inside	Outside	Total	Inside	Outside	Total
<i>Chelodina longicollis</i>	211	361	572	21	77	98
<i>Tiliqua rugosa</i>	251	106	357	5	0	5
<i>Pogona barbata</i>	81	22	103	1	2	3
<i>Tiliqua scincoides</i>	10	6	16	1	0	1
<i>Pseudonaja textilis</i>	3	0	3	1	0	1
<i>Egernia cunninghami</i>	1	0	1	0	0	0

those trapped in ponds in the vicinity of the fence. Size-frequency distributions were compared with a chi-square test with the null hypothesis of equal frequency between samples for all size class groups. We followed the overall test with a series of chi-square tests to determine where specific differences existed. We lowered significance values for this series of comparisons to  $\alpha < 0.005$  using the Dunn-Sidak correction for avoiding false positives arising from compounding error. We performed chi-square tests to examine differences in sex ratios between turtles encountered at the fence versus ponds, and the inside versus outside of the fence.

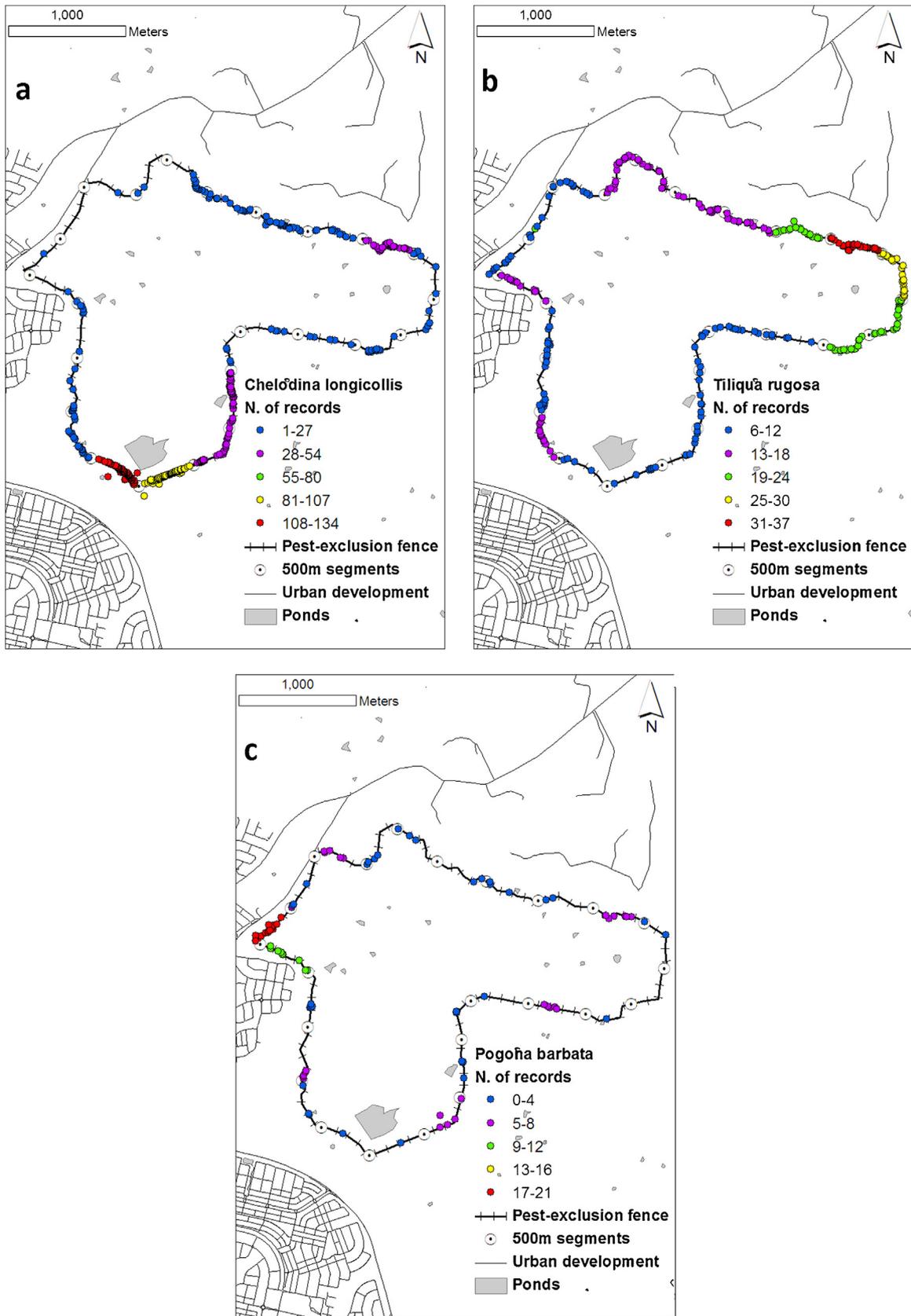
To estimate the magnitude of fence bycatch on turtles at the population level, we estimated population size (mean  $\pm$  SE) within the five sampled nature reserve ponds using the Horvitz-Thompson type estimator  $N = (n/p)$ , where  $N$  is the estimated population abundance,  $n$  is the number of turtles captured and  $p$  is the estimated recapture probability (Seber 1982). We used recapture probabilities derived from previous studies using similar trapping protocols in the same study system (Roe, Rees, & Georges 2011). We then calculated population density (number/ha) in the five surveyed ponds and extrapolated density estimates to all ponds within the fenced area to obtain a within-reserve population size estimate.

## Results

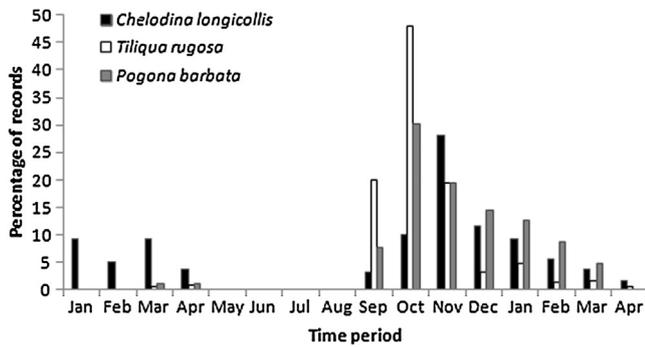
### Encounters and mortalities

We encountered six species during fence surveys, totalling 1052 reptile encounters, with *C. longicollis* the most common species followed by shingleback lizards (*Tiliqua rugosa*), eastern bearded dragon lizards (*Pogona barbata*), blue tongue skinks (*Tiliqua scincoides*), eastern brown snakes (*Pseudonaja textilis*), and Cunningham's skinks (*Egernia cunninghami*) (Table 1).

Of 108 dead animals encountered, the majority were *C. longicollis* (90.7%), followed by *T. rugosa* (4.6%), *P. barbata* (2.8%), *T. scincoides* (0.9%) and *P. textilis* (0.9%). The most common cause of death in *C. longicollis* was overheating ( $n=68$ ; inside = 10, outside = 58), followed by fox depredation ( $n=18$ ; inside = 0, outside = 18), vehicular collision ( $n=10$ ; inside = 10, outside = 0), and entanglement ( $n=2$ ; inside = 1, outside = 1). Overheated turtles had signs of skin sunburn, mouth opened and neck and hyoid extended. Turtles depredated by foxes had the limbs and/or head eaten, though these signs may also suggest scavenging by foxes. Five *T. rugosa*, one *P. barbata* and one *T. scincoides*, all adult individuals, died from overheating; one *P. textilis* died from entanglement; and two *P. barbata* died from electrocution. We recaptured several *C. longicollis* along the fence. Of the 71 *C. longicollis* recaptured alive, 68% were captured twice, 27% three times, 4% four times, and 1% five times. Fifty *C. longicollis* were recaptured and eventually found dead (70% on the second recapture, 16% on the third recapture, 12% on the fourth recapture, and 2% on the sixth recapture), indicating persistent attempts to navigate past the fence.



**Fig. 2.** Spatial distribution of encounters for (a) *Chelodina longicollis*, (b) *Tiliqua rugosa*, and (c) *Pogona barbata* along the pest-exclusion fence at Mulligans Flat Nature Reserve, Australian Capital Territory, Australia. Each point represents a unique animal encounter, color-coded to reflect the number of encounters occurring within the 23 fence segments.



**Fig. 3.** Temporal patterns of reptiles records from January 2012 to April 2013 in the pest-exclusion fence at Mulligans Flat Nature Reserve, Australian Capital Territory, Australia.

#### Spatial correlates of encounters

Encounters were not equally distributed among the fence segments for any species (*C. longicollis*:  $X^2 = 1158.6$ ,  $df = 22$ ,  $p < 0.001$ ; *T. rugosa*:  $X^2 = 75.1$ ,  $df = 22$ ,  $p < 0.001$ ; *P. barbata*:  $X^2 = 89.7$ ,  $df = 22$ ,  $p < 0.001$ ) (Fig. 2). *Chelodina longicollis* encounters were positively correlated with total wetland area and negatively correlated with urban development area, explaining 51% of the variation ( $F_{2,19} = 9.90$ ,  $r^2 = 0.51$ ,  $p < 0.001$ ) (Predictor variables: Wetland area:  $\text{Beta} = 0.55$ ,  $p = 0.003$ ; Urban development area:  $\text{Beta} = -0.37$ ,  $p = 0.035$ ). However, *T. rugosa* ( $F_{4,18} = 2.34$ ,  $r^2 = 0.34$ ,  $p = 0.09$ ) and *P. barbata* ( $F_{4,17} = 2.77$ ,  $r^2 = 0.39$ ,  $p = 0.06$ ) encounters were not correlated with any of the measured land cover variables.

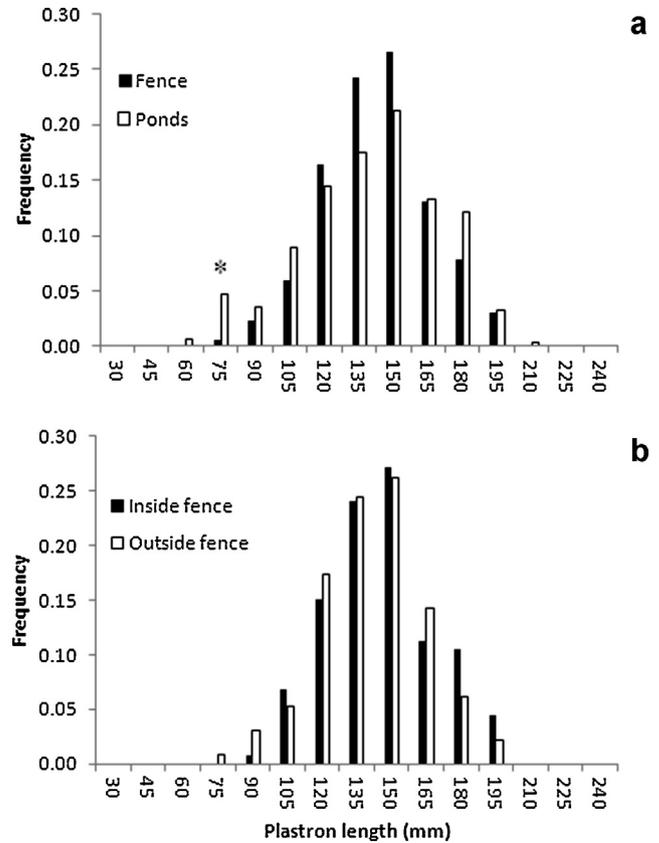
#### Temporal correlates of encounters

Encounters were not equally distributed among months (*C. longicollis*:  $X^2 = 677.2$ ,  $df = 15$ ,  $p < 0.001$ ; *T. rugosa*:  $X^2 = 1193.2$ ,  $df = 13$ ,  $p < 0.001$ ; *P. barbata*:  $X^2 = 158.9$ ,  $df = 13$ ,  $p < 0.001$ ) (Fig. 3). *Chelodina longicollis* encounters were positively correlated with daily solar exposure and rainfall, and negatively correlated with daily maximum temperature, explaining 70% of the variation ( $F_{3,20} = 15.72$ ,  $r^2 = 0.70$ ,  $p < 0.001$ ) (Predictor variables: daily solar exposure:  $\text{Beta} = 1.63$ ,  $p < 0.001$ ; daily maximum temperature:  $\text{Beta} = -1.32$ ,  $p < 0.001$ ; rainfall:  $\text{Beta} = 0.32$ ,  $p = 0.020$ ). *Tiliqua rugosa* encounters were positively correlated with daily solar exposure, and negatively correlated with daily maximum temperature, explaining 48% of the variation ( $F_{2,26} = 11.80$ ,  $r^2 = 0.48$ ,  $p < 0.001$ ) (Predictor variables: daily solar exposure:  $\text{Beta} = 1.21$ ,  $p < 0.001$ ; daily maximum temperature:  $\text{Beta} = -0.73$ ,  $p = 0.014$ ). *Pogona barbata* encounters were positively correlated with daily solar exposure and rainfall, explaining 70% of the variation ( $F_{2,26} = 31.10$ ,  $r^2 = 0.70$ ,  $p < 0.001$ ) (Predictor variables: daily solar exposure:  $\text{Beta} = 0.79$ ,  $p < 0.001$ ; rainfall:  $\text{Beta} = 0.29$ ,  $p = 0.012$ ).

Our logistic regression model identified location ( $p = 0.002$ ), season ( $p = 0.004$ ), daily maximum temperature ( $p < 0.001$ ), rainfall ( $p = 0.001$ ), and daily solar exposure ( $p < 0.001$ ) as predictors of *C. longicollis* mortality along the fence (Table 2). Turtles had a higher chance of dying on the outside of the fence, and a lower probability of dying during Autumn and Spring. Mortalities were positively correlated with daily maximum temperature and daily solar exposure, but negatively correlated with rainfall (Table 2).

#### Size-frequency distributions

Size-frequency distributions differed between turtles encountered at the fence and pond captures (overall  $X^2 = 27.68$ ,  $df = 10$ ,  $p = 0.002$ ), but the only significant disparity occurred within the



**Fig. 4.** Size-frequency distributions of *Chelodina longicollis* (a) captured in ponds compared to those moving along a pest-exclusion fence, and (b) those that were found inside of the pest-exclusion fence compared to those outside at Mulligans Flat Nature Reserve, Australian Capital Territory, Australia. Asterisk indicates size class that differed in frequency of occurrence between sites, adjusted to  $\alpha < 0.005$ .

60.1–75 mm PL size class ( $X^2 = 11.5$ ,  $df = 1$ ,  $p < 0.005$ ) (Fig. 4). Female encounters at the fence were similar to ponds, while males were 1.6 times more commonly found at the fence compared to pond captures ( $X^2 = 13.54$ ,  $df = 2$ ,  $p = 0.001$ ). Frequency of encounters for turtles on the inside and outside of the fence did not differ for any demographic group (size-frequency distributions:  $X^2 = 7.81$ ,  $df = 8$ ,  $p = 0.45$ ), or sex and age class ( $X^2 = 0.56$ ,  $df = 2$ ,  $p = 0.75$ ) (Fig. 4).

#### Magnitude of disruption

Turtle population density within the five sampled ponds was 77 individuals/ha (range: 56–125), which extrapolates to an estimated population size of 641 (range: 466–1041) within the fenced reserve. Movements were disrupted for 20.9% (range: 12.9–28.8%) of the reserve population, with 3.3% (range: 2.0–4.5%) of the population killed along the fence.

#### Discussion

To our knowledge, this is the first quantification of the impact of a pest-exclusion fence on a reptile community, and one of the most detailed studies on the impact of fences in non-target species. Our main findings relative to feral pest management were that (1) several non-target native wildlife species had their movements restricted by the fence; (2) impacts were most severe for turtles, both in fragmenting local populations by disrupting movements and causing mortality at levels likely to threaten long-term population viability; and (3) encounters and mortalities were predictable in space and time, which would allow for species-specific

**Table 2**  
Logistic regression model base for timing of *Chelodina longicollis* deaths along the pest-exclusion fence at Mulligans Flat Nature Reserve, Australian Capital Territory, Australia.

	B	S.E.	Wald	df	Sig.	Exp(B)	95% C.I. for EXP(B)	
							Lower	Upper
Location (1)	1.650	0.536	9.483	1	0.002	5.209	1.822	14.894
Season			15.442	4	0.004			
Season (1)	−13.236	4164.478	0.000	1	0.997	0.000	0.000	
Season (2)	−5.543	1.411	15.435	1	0.000	0.004	0.000	0.062
Season (3)	−5.482	1.483	13.654	1	0.000	0.004	0.000	0.076
Season (4)	−19.612	6001.747	0.000	1	0.997	0.000	0.000	
SEX			2.627	2	0.269			
SEX (1)	−0.080	0.537	0.022	1	0.881	0.923	0.322	2.644
SEX (2)	0.520	0.545	0.909	1	0.340	1.682	0.578	4.897
CL <sup>a</sup>	−0.027	0.082	0.107	1	0.744	0.974	0.829	1.143
DMT <sup>b</sup>	0.421	0.117	12.949	1	0.000	1.523	1.211	1.915
Rain <sup>c</sup>	−0.088	0.026	11.329	1	0.001	0.916	0.871	0.964
DSE <sup>d</sup>	0.665	0.156	18.122	1	0.000	1.944	1.432	2.641
Constant	−25.880	5.519	21.988	1	0.000	0.000		

<sup>a</sup> CL, carapace length.

<sup>b</sup> DMT, daily maximum temperature.

<sup>c</sup> Rain, rainfall accumulated.

<sup>d</sup> DSE, daily solar exposure.

management of fence bycatch. Together, these findings suggest that current and future fence designs that do not consider the ecology of non-target species can lead to negative consequences for populations of native wildlife.

#### General impacts on reptiles

The current design of the pest-exclusion fence precluded movements of native wildlife of several species of large lizards and snakes, but impacts were most severe for turtles. While it is possible that such variation reflects relative population sizes of species in the area, it is more likely that the placement of the fence interferes with differences in important species-specific movement routes. One might expect that a terrestrial fence would have little impact on a nominally aquatic species of turtle. However, *C. longicollis* moves overland for nesting, to locate terrestrial refugia for estivation, and during movements among wetlands (Kennett et al. 2009; Roe & Georges 2007; Roe, Brinton, & Georges 2009). The large proportion of the population encountered at the fence representing all demographic groups from regional wetlands underscores the importance of such movements for *C. longicollis* population. A structure blocking their path would thus disrupt natural behaviors, potentially reducing survival directly or restricting access to critical resources. The lizards and snake species found by the fence occasionally move large distances, but in general they exhibit small home ranges (Fergusson & Algar 1986; Price-Rees, Brown, & Shine 2012; Whitaker & Shine 2003). Considering the relatively low number of lizard and snake encounters and mortalities, it is likely that the fence is less restrictive of their movements compared to turtles.

Most of the turtle records were on the outside of the fence clustered in a few regions along the fence perimeter (Fig. 2), suggesting turtles were attempting to access habitats and resources within the reserve. Previous research in this system revealed that, before the fence was built, drought conditions forced many turtles to move from the nature reserve into the larger suburban ponds for refuge (Rees, Roe, & Georges 2009; Roe, Rees, & Georges 2011), opposite the predominant direction we observed in our study. During drought, water levels in the nature reserve ponds tend to fluctuate more and experienced greater drying than the suburban ponds (Rees, Roe, & Georges 2009). The attempted movements back into the reserve are likely in response to the reflooding of wetlands following considerably more rainfall during and just prior to the current study (study period rainfall: 965 mm, Australian Bureau

of Meteorology) compared to previous years (649 mm, Australian Bureau of Meteorology) (Rees, Roe, & Georges 2009; Roe, Rees, & Georges 2011). Such two-way movements between permanent and ephemeral water bodies are key elements affecting survival and regional carrying capacity of *C. longicollis* in response to wet-dry cycles (Kennett et al. 2009; Rees, Roe, & Georges 2009; Roe & Georges 2007; Roe, Brinton, & Georges 2009; Roe, Rees, & Georges 2011), and the fence is completely disrupting this dynamic. Any animals that require extensive movement through the landscape would be severely impacted by exclusion fencing, as such barriers can disrupt population and metapopulation dynamics, gene flow, and population rescue from stochastic events (Boonstra & Krebs 1977; Epps et al. 2005; Flesch et al. 2010; Krebs, Keller, & Tamarin 1969; Lesica & Allendorf 1995; Marsh & Trenham 2001).

In addition to being an impenetrable barrier, the fence also caused considerable indirect mortality in *C. longicollis*. Overheating was the presumed primary cause of death, as areas around the fence offered little refuge for turtles, which can result in rapid overheating when exposed to unsuitable conditions. Critical thermal maximum for turtles occurs at around 40 °C (Hutchison, Vinegar, & Kosh 1966), and prolonged (>1 h) central body temperatures > 38 °C can lead to dehydration and eventual death (Lagarde et al. 2012). As would be expected, days with higher temperatures and solar index, and little to no rainfall resulted in the highest probability of turtle mortality. Entanglement was not a major cause of mortality in our study site, being responsible for only two cases in turtles and one in snakes. Mesh sizes in Australia (30 mm hexagonal netting) are mainly framed to protect native species against large predators, such as cats, rabbits and foxes, while mesh sizes in New Zealand are smaller (6 mm × 50 mm), as small mammals (house mice) are the main threat to native species (Burns, Innes, & Day 2012; Moseby & Read 2006). A larger mesh size would probably not be desirable as they would allow juvenile rabbits to pass through (Moseby & Read 2006). A possible reduction in mesh somewhere between these two measurements above could help to prevent entanglement, though this would need to be experimentally tested.

We observed that many turtles were recaptured at the fence several times, only to be eventually found dead, demonstrating that they repeatedly attempted the movement despite the obstacle. We conservatively estimate that the fence caused mortality in 3.3% (2.0–4.5%) of the reserve turtle population during the period of our study. These numbers are likely an underestimate of the actual impact of the fence on turtles at the population-level, considering many dead turtles would have gone undetected by observers

due to scavenging and observer bias. Moreover, many turtles found outside the fence were likely part of the reserve population in the process of migrating back to the reserve following the end of drought, as *C. longicollis* populations here and elsewhere move among several water bodies with groups of permanent and temporary wetlands harboring the most relevant population unit (Roe, Brinton, & Georges 2009; Roe, Rees, & Georges 2011). Even with the uncertainty of these estimates, such mortality (if chronic) is likely to threaten long-term population viability in turtles (Brooks, Brown, & Galbraith 1991; Congdon, Dunham, & Van Loben Sels 1993; Heppell 1998). While there may be some benefits for turtles within the pest-exclusion zone via relief from fox predation on nests (Spencer 2002; Thompson 1983), the number of adult deaths observed here would likely still drive population declines (Congdon, Dunham, & Van Loben Sels 1993) and limit recovery following stochastic events such as drought. Ironically, fox depredation on turtles was high along the fence designed to protect wildlife from the foxes.

#### Management, mitigation and non-target species

Our investigation demonstrates that fences can affect non-target species and management is going to be most effective when the behavior and population dynamics of native species in the system are considered during threat/risk assessments. Establishing pest-proof fences in National Parks (e.g. Booderee NP, see Roe & Georges 2007) or other protected areas set aside for native species conservation and maintenance of natural system functioning can be questioned in some cases. Perhaps such fences would be best established adjacent to protected areas of high value where they will, through concurrent habitat improvement, raise conservation value more generally and achieve benefits beyond feral predator control. Also, the rationale for the fence establishment should be evaluated in terms of balancing costs and benefits for target and non-target species. We acknowledge that conservation fences can be the only option for the conservation of some native species due to the enormous effect of introduced mammalian predators (Dickman 2012; Dowding & Murphy 2001; Hayward & Kerley 2009). Perhaps in cases where fences are constructed for *in situ* protection of remnant populations of endangered species and time is a limiting factor to halt the threatening processes, one could limit costs to non-target species by fully implementing mitigation strategies to reduce the impact on them. Nonetheless, in the case of reintroduction of locally extinct fauna, managers still have time to consider and balance costs and benefits. In such situations, fences should be erected only in areas where they pose a minimum risk to non-target species.

There is a current debate in New Zealand, a country with tradition in the use of conservation fences, over whether predator-proof fences are the best management option to conserve native species against feral animals, with some suggesting conservation fences are not cost-effective over the long-term, and that the fence projects frequently do not achieve the goals of species conservation and recovery (Scofield, Cullen, & Wang 2011). Others proclaim the benefits of predator-proof fences and cite several examples of species recovery and feral pest exclusion, pointing out that fence projects can also provide social and education benefits, an infusion of funding from non-traditional sources, and powerful advocacy on behalf of other conservation initiatives (Innes et al. 2012). Adding to such discussions, Norbury et al. (in press) recently evaluated cost-effectiveness of pest fencing *versus* pest trapping in New Zealand, finding that predator proof-fences are most cost-effective for areas below 1 ha, semi-permeable (“leaky”) fences are most cost-effective for 1–219 ha, and trapping is the most cost-effective for areas above 219 ha (Norbury et al. in press).

Conservation fences are clearly an effective management practice in many circumstances (Dickman 2012; Hayward & Kerley

2009), including some cases in which fences were erected specifically to protect turtles and their habitats (Doupé et al. 2009; Kuchling 2000). For a successful implementation of conservation fences, managers require specific information to guide them in fence construction and maintenance. In addition, managers need to prioritize sites for localized mitigation measures (i.e. hotspots) and periods when threat rates are highest (i.e. hot moments) as such strategies reduce the risk of mortality and are essential to the long-term survival of native species in challenging landscapes (Beaudry, deMaynadier, & Hunter 2008; Beaudry, deMaynadier, & Hunter 2010). In our system, the fence layout should avoid bisecting travel routes among wetlands, and based on the movement capacity of *C. longicollis*, this should include distant wetlands up to 1.5 km away (Roe & Georges 2007; Roe, Brinton, & Georges 2009). Where potential disruptions to movements cannot be avoided, mitigation to facilitate movements and avoid mortality could be applied at targeted times and locations, in accordance with our analysis, including: (1) manually assisting animals across the fence at hot moments for mortalities (perhaps using volunteers), (2) conducting a poison bait program for foxes (Long & Robley 2004) along animal hotspots, (3) regularly mowing the vegetation close to the fence and covering the puddles with gravel to increase driver visibility and reduce the chances of vehicular mortality, and (4) building aquatic under-passages (Long & Robley 2004) or size-specific mesh (Roe & Georges 2007) along animal hotspots that facilitate non-target animal movements while excluding pests. However, the purpose of our study was not to test effective solutions. Encouragingly, managers have been presented with other threats to turtles, such as fisheries bycatch and roads (Brewer et al. 2006; Gibbs & Shriver 2002; Lewison & Crowder 2007) and through careful study, progress has been made toward implementing successful solutions, at least on local scales (Beaudry, deMaynadier, & Hunter 2008; Brewer et al. 2006; Dodd, Barichivich, & Smith 2004; Lewison & Crowder 2007). We are not suggesting that fencing is a problem of equal magnitude to roads and fisheries bycatch, but it is nonetheless a parallel scenario that warrants action, and it is not an intractable problem. While the decisions of whether or not to build a fence and which mitigation practices to employ are likely to be complex and unique to each fence project, our study nonetheless provides an example of the process to help inform managers of potential conflicts and solutions.

#### Conclusion

Although we focused on a single conservation fence, we believe that the impact of fences on non-target species is widespread in Australia and elsewhere. In Australia alone, there are at least 37 pest-exclusion fences projects enclosing 136,342 ha (Dickman 2012; Long & Robley 2004), and other large fence enterprises include the dingo fence covering 5631 km (Allen & Sparkes 2001), the rabbit proof fence extending 3256 km (Prober & Smith 2009), and the emu proof State Barrier Fences extending 1170 km (Pickard 2007b). Additionally, there are countless other undocumented kilometers of rural fencing to protect agriculture and livestock, but the current length is unknown (Pickard 2007a). Anecdotal evidence on the mortality of turtles in sheep farms was brought to us by farmers in the region, and we have observed turtle entanglement in such fencing previously (J.H.R. and A.G., personal communication), in addition to reports of turtle death against rabbit fencing in Victoria, Australia (Anon. 1941). Pest-exclusion fencing is common in other countries as well, including at least 31 such fences enclosing 9160 ha in New Zealand (Burns, Innes, & Day 2012; Scofield, Cullen, & Wang 2011) and Zimbabwe, where veterinary fences extend for 2250 km (Taylor & Martin 1987). All of these types of fences have the potential to exclude and kill native vertebrates, and

continued implementation of this widespread and popular conservation and management practice should be critically examined to avoid further collateral damage.

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