CSIRO PUBLISHING

Australian Journal of Zoology, 2017, **65**, 203–216
https://doi.org/10.1071/ZO17041

Decline of the dasyurid marsupial *Antechinus minimus maritimus* in south-east Australia: implications for recovery and management under a drying climate

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Abstract. Our understanding of recent extinctions and declines of Australian mammals is poor, particularly where there is a paucity of data to quantify change. The swamp antechinus (*A. m. maritimus*) has a fragmented, coastal distribution in south-east Australia. Although long-term studies (1975–2007) of this vulnerable species were conducted in the eastern Otways, its current status was unclear. We assessed the success of live trapping and camera trapping (2013–17) at 42 sites, 19 where the species was trapped previously. Between 2013 and 2015 *A. m. maritimus* was recorded at only 6 sites ($n = 8$), but at none in 2016–17. Assessment of long-term changes found that high-density populations occurred after above-average rainfall, and both low- and high-density populations collapsed after wildfire, after low rainfall, and in fragmented habitat. The species may now be restricted to very small populations in refuges such as coastal dunes, and predicted low rainfall and increased burning frequency pose major threats to the species' survival. Recovery is unlikely without targeted management, including predator control and protection from inappropriate fire regimes and habitat fragmentation. If similar declines have been experienced across the species' range, prevention of extinction of the species will require similar management strategies.

Additional keywords: fire, population, rainfall, swamp antechinus.

Received 2 August 2017, accepted 1 November 2017, published online 22 November 2017

Introduction

Since European settlement ground-dwelling Australian mammals have experienced one of the worst extinction rates (Burbidge and McKenzie 1989; Johnson 2006; McKenzie *et al.* 2007; Burbidge *et al.* 2008; Woinarski *et al.* 2014, 2015; Wayne *et al.* 2017) in the world. Many extant species have declined significantly and are now threatened; however, we have a poor understanding of how or why they have declined, especially cryptic species such as the smaller dasyurid marsupials (Menkhorst 1995; Wilson *et al.* 2003; Bilney *et al.* 2010). In total, 24% of smaller dasyurids (<500 g) were classified as vulnerable, endangered or data deficient in 1996 (Maxwell *et al.* 1996; Wilson *et al.* 2003). By 2015, 31% were classified as vulnerable, endangered or near threatened; the increase in near threatened resulting from more adequate information (Woinarski *et al.* 2014). It is recognised that effective monitoring to quantify substantial or significant declines in appropriate timeframes is required to inform management decisions and conservation activities (Burgman *et al.* 2012; Lindenmayer *et al.* 2012, 2014; Varcoe 2012). However, long-term monitoring programs to quantify population changes and threats for this group of small mammals have been limited.

The swamp antechinus (*Antechinus minimus maritimus*) is a small terrestrial dasyurid that has a restricted, disjunct distribution. It occurs predominantly along coastal south-eastern Australia from south-eastern South Australia to Wilsons Promontory, Victoria, and near offshore islands (Menkhorst 1995; Bachmann and van Weenen 2001; Wilson *et al.* 2001). *A. m. maritimus* favours damp habitat, particularly dense heathlands and woodlands, tussock grasslands, sedgelands and gullies (Menkhorst 1995), often in landscape settings with little exposure to the sun (Wilson *et al.* 2001; Gibson *et al.* 2004). Habitat suitability modelling indicates that only a small proportion (10–15%) of the distributional extent provides high-quality habitat for this subspecies (Wilson *et al.* 2001; Gibson *et al.* 2004; Magnusdottir *et al.* 2008). Much of *A. m. maritimus* habitat has been cleared and swamps drained, and fragmented populations may be at ongoing high risk of local extinction (Bachmann and van Weenen 2001; Wilson *et al.* 2001; van Weenen and Menkhorst 2008; Wilson and Bachmann 2008). Despite these threats the species has until recently been considered as at Lower Risk (Maxwell *et al.* 1996; Wilson *et al.* 2003).

Breeding in *A. m. maritimus* is seasonal with a distinct breeding period from winter to spring. Mating is highly

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synchronised, males die after mating and females give birth to 6–8 young (Wilson and Bourne 1984; Wilson 1986; Sale *et al.* 2006; Magnusdottir *et al.* 2008). Its diet includes a wide range of invertebrates, particularly moth larvae and beetles, and some small vertebrates and seeds (Allison *et al.* 2006; Sale *et al.* 2006).

A. m. maritimus prefers structurally complex late-successional habitat. This preferred habitat is part of a wider landscape that may be impacted by controlled burns and wildfire (Wilson *et al.* 1986; Moro 1991; Wilson 1996; Gibson *et al.* 2004). In Victoria local extinctions of populations have been recorded after severe wildfire with no recolonisation in 33 years subsequent to fire. However, some populations survived in coastal sand dunes where burns were patchy (Wilson 1996; Wilson *et al.* 2001). There is thus a need to consider how this species will be impacted under the introduction of altered fire management regimes involving increased fuel reduction burning, arising from the 2009 Victorian Bushfires Royal Commission (Parliament of Victoria 2010; Department of Environment and Primary Industries 2014).

Significant population declines of the species have been recorded during periods of below-average rainfall and drought (Magnusdottir *et al.* 2008; Sale *et al.* 2008), especially during the 'millennium drought' (1996–2010) where much of the south-east of Australia experienced persistent drought (CSIRO and Australian Bureau of Meteorology 2015).

Climate change predictions of declining rainfall and extended drought (CSIRO and Australian Bureau of Meteorology 2012, 2014) are likely to pose significant threats to this vulnerable species, especially in fragmented ecosystems. Other potential threats to *A. m. maritimus* include long-term vegetation degradation due to the plant pathogen *Phytophthora cinnamomi* (Laidlaw and Wilson 2006; Annett 2008) and predation by the red fox and cats (Wilson *et al.* 2001; van Weenen and Menkhorst 2008; Woinarski *et al.* 2014).

It is important to examine how changes to fire frequency, together with interactions with lower rainfall and extended drought will impact the species at local and regional scales. An evaluation of long-term data can inform how changes have impacted, and are likely to impact mammal populations and how management needs to be adaptive to such changes.

Although long-term research investigating the reproduction and population ecology of *A. m. maritimus* was conducted since 1975 in the eastern Otways, these studies ceased in 2007 (Wilson *et al.* 1986, 2001; Aberton 1996; Reichl 1997; Hanley 1999; Gibson *et al.* 2004; Magnusdottir *et al.* 2008; Sale *et al.* 2008). In 2013 it was evident that the status of the species in the region was unclear and required reassessment, leading to the current study. The aims of this study were thus to: (1) investigate the current distribution and status of *A. m. maritimus* in the eastern Otways; (2) compare the current distribution and abundance to previous long-term records; and (3) consider the impacts of threats (habitat fragmentation and degradation, too frequent fire, introduced predators, and climate change) and to provide information to support effective monitoring, management and recovery programs for the species throughout its range.

Materials and methods

Study area

The study area in the eastern Otway Ranges is 100 km south-west of Melbourne, Victoria (Fig. 1). The 22 000-ha area is predominantly public land, including the Great Otway National Park and the Anglesea Heath, which is recognised for its high biodiversity values (McMahon and Brighton 2002). The vegetation communities comprise a diverse mosaic of predominantly eucalypt forests, woodlands and heathlands, interspersed with dense wet shrublands (Land Conservation Council Victoria 1985). Between 1961 and 2016 ~7500 ha of this area was leased by Alcoa of Australia Pty Ltd for the purpose of brown coal extraction (Fig. 1).

The most extensive recent fire in the area, the 'Ash Wednesday' wildfire (1983), burnt 40 000 ha and left few unburnt small patches of vegetation in the study area (Wilson *et al.* 1990). Fuel reduction burns have been undertaken since 1986 and an increase in fuel reduction burning is being implemented as a result of recommendations arising from the 2009 Victorian Bushfires Royal Commission (Parliament of Victoria 2010; Coates *et al.* 2013; Department of Environment and Primary Industries 2014). Regular fox baiting has been undertaken since 2005 (Parks Victoria and Department of Sustainability and Environment 2009).

Assessment of current distribution and abundance of *A. m. maritimus* (2013–17)

We re-established trapping at sites that were originally studied between 1975 and 2007. Live-trapping was undertaken between May 2013 and April 2017 at 42 sites, 19 where the species had been trapped previously, and the other sites as potential locations

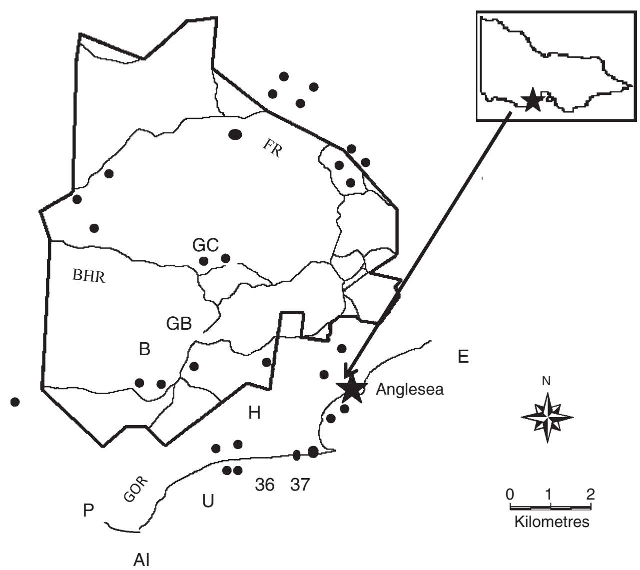

Fig. 1. Study area in the eastern Otways showing all survey locations for *A. m. maritimus* including long-term trapping sites: Harvey (H), Batsons (B), Bald Hills Grid B (GB), Grid C (GC), Urquhart (U), Eumeralla (E), Aireys Inlet (AI), Sanddunes (36, 37), and Painkalac (P). Great Ocean Road (GOR), Bald Hills Road (BHR), Forest Road (FR), and sites surveyed in 2013–17 (●) are also shown. The thick black line shows the Alcoa Lease Boundary.

in what was deemed favourable *A. m. maritimus* habitat (Gibson *et al*. 2004). Trapping was conducted annually (1–3 times per site) over a total of 6378 trap-nights at 21 sites in 2013–14, 28 sites in 2015, 13 sites in 2016, and at 8 sites in 2017 (Supplementary Table S1). Camera trapping was conducted at one site in 2013–14, at 20 sites in 2015, at five sites in 2016 and six sites in 2017 (Supplementary Table S2).

Trapping was undertaken using Elliott traps (325 × 90 × 100 mm) baited with a mixture of rolled oats, honey and peanut butter. At most sites 30 Elliott traps were set in transect formation in lines of 10 at 10–15-m intervals, except at Bald Hills (Grid B) where 100 traps were set in a 10 × 10 configuration, and sites where the habitat patch was very small, where only 4–10 traps were set. Traps were set for three nights. Species, sex, reproductive condition, body weight (g), and body measurements, including head length (mm), head–body length (mm), tail length (mm) and pes length (mm), were recorded for each individual animal. An estimation of age was made for each individual based on weight, body measurements and reproductive condition. Captured *A. m. minimus* individuals were identified or marked with an ear tag before release at point of capture.

At each site two cameras (Reconyx Hyperfire HC600 Passive Infrared) were positioned ~100 m apart, for 1–5 weeks. Cameras were set to high sensitivity with a 15-s interval between triggers and three images were taken per trigger. Bait lures were not employed in order to avoid bias towards any specific species. Cameras were secured vertically 0.2–1 m from ground level and dense vegetation was cleared from the field of view. Photographs of individuals were identified to species level by using reference photographs from the area (BW, unpubl. data) and species descriptions (Van Dyck and Strahan 2008; Van Dyck *et al*. 2013).

Trapping and camera data analyses

Although mark–recapture and other analyses have been used to derive estimates of populations (number of individuals known to be alive; density) for *A. m. maritimus* at particular sites (Wilson *et al*. 1986, 1990, 2001; Annett 2008; Magnusdottir *et al*. 2008; Sale *et al*. 2008) this precise population information was not available at several important survey sites or for data series at sites. To ensure consistent comparison across years we used trapping rate as a standard measure of abundance: trap success = (total captures/total number of nights) × 100.

For camera trapping data the total number of independent captures (visits) was used to calculate trapping success at each site by: trap success = (total captures/total number of nights) × 100. An independent capture (visit) was defined as a 30-min interval between images of the same species (De Bondi *et al*. 2010; Claridge *et al*. 2010).

Assessment of long-term abundance trends at repeat trapping sites (1975–2017)

Long-term data were available for *A. m. maritimus* captures at 10 sites in the eastern Otways in heathy woodland, sand heathland, coastal headland scrub, coastal dune scrub and estuarine wetland (Table 1, Fig. 1). Trapping methods employed followed the same general methods described above for the assessment of current distribution and abundance (2013–17) including setting of traps for three nights.

Trapping was conducted at the Harvey site on 26 occasions between 1975 and 1979, and from March 1980 to September 1981 on a monthly basis (Tables 1 and 2). In 1983 the site was severely burnt by wildfire and trapping was then conducted bimonthly from May 1984 to February 1985 and then seasonally from winter 1986 to autumn 1988, and in 1995 and 2002. Recent survey trapping was conducted in June 2014, May 2015, and April 2017, providing trapping data over a 42-year period, and camera trapping in May 2015 and April 2017.

At the Bald Hills site (Table 1) trapping was undertaken on two grids (Grid B, Grid C) from 1988 to 2005: Grid B on 31 occasions, Grid C on 18 occasions (Tables 1 and 2). In the recent assessments survey trapping was conducted in June and October 2013, April 2015, February 2016 and March 2017, providing trapping data over a 29-year period. Camera trapping was completed in May 2015 and March 2017.

Trapping at the Urquhart site (Table 1) was undertaken on 35 occasions between 1998 and 2007 (Table 2). Recent trapping surveys were completed in October 2013, September 2014, May 2015, March 2017, and camera trapping September–October 2014, May 2015 and February 2017.

At the Eumeralla site (Table 1) live-trapping trapping was conducted from 2004 to 2015 on 19 occasions (Homan 2017) and recently in February 2016, together with camera trapping.

Trapping at the Batson site was first undertaken monthly from January to June 1981 (Tables 1 and 2). Annual trapping was conducted following the 1983 wildfire between 1984 and 1991 and following trial *A. m. maritimus* reintroductions ($n = 10$ animals per trial) in 1992, 1993 and 1994 (Aberton 1996). Subsequent survey trapping was undertaken in 1998, 1999, 2001, 2004 (Table 2). Recent trapping was undertaken in March 2016, providing trapping data over a 36-year period.

Trapping at Aireys Inlet took place from March 1999 until 2003 at sites located along cliff-tops in small patches of coastal headland scrub (Tables 1 and 2). Recent trapping was conducted in June 2014, May 2015 and April 2017.

Trapping surveys at two coastal dune sites (36, 37) were implemented following the 1983 wildfire, with trapping taking place on 13 occasions between 1983 and 2004 (Tables 1 and 2). Survey trapping and camera trapping were conducted recently in June 2015 and March 2017.

Trapping occurred between 1999 and 2003 at the Painkalac Creek site (Tables 1 and 2), on 15 occasions, and recent survey trapping was completed in June 2015 and April 2017.

The abundance (trap success) results for the long-term data were calculated based on the maximum success (1975–2007) and (2013–17). Trends were categorised as either severe decrease (>80% reduction in abundance), decline (30–80% reduction in abundance), slight decline (0–30% reduction in abundance), or increase over the course of the monitoring period (Wayne *et al*. 2017).

Fire history and rainfall patterns

The fire history (time since last burnt; number, area and type of burn) at individual *A. m. maritimus* sites was obtained; and the fire history of the major Ecological Vegetation Divisions (EVDs) (total area burnt and the percentage area burnt for each EVD) was

Table 1. Summary description of long-term *A. m. maritimus* study sites

EVC, Ecological Vegetation Classes: standard unit for classifying vegetation types in Victoria, based on floristics, lifeforms and ecological characteristics

Site	EVC	Description, dominant plant species
Harvey	Heathy woodlands	Inland south valley. *Eucalyptus obliqua* (messmate stringybark), dense wet areas of *Leptospermum continentale* (prickly tea-tree) 4 m; low heath *Banksia marginata* (silver banksia), *Danthonia penicillata*, *Dillwynia glaberrima* (smooth parrot-pea), *Epacris impressa* (common heath), *Gahnia radula* (thatch saw-sedge), *Pteridium esculentum* (austral bracken), *L. myrsinoides* (silky tea-tree), *Xanthorrhoea australis* (austral grass-tree).
Batson	Heathy woodland, lowland forest	Inland south slope. *L. juniperinum*, *E. impressa*, *Amperea xiphoclada* (broom spurge), *D. glaberrima*, *G. radula*, *P. esculentum*, *Acacia longifolia* (sallow wattle), *Opercularia varia* (variable stinkwort), *Dichondra repens* (kidneyweed).
Bald Hills B	Heathy woodland, lowland forest	Inland south slope. Low canopy *Eucalyptus willissii* (shining peppermint), mid-storey *L. myrsinoides*, *X. australis*, *L. continentale*, *Monotoca scoparia* (heath broom), *D. glaberrima*, *E. impressa*.
Bald Hills C	Heathy woodland riparian scrub	Inland south slope, low upper storey *E. willissii*, mid-storey *L. myrsinoides*, *X. australis*, *L. continentale*, *M. scoparia*, *D. glaberrima*, *E. impressa*.
Urquhart	Sand heathland	Coastal south slope. Upper storey *E. obliqua*, *L. continentale*, mid–understorey *G. radula*, *L. myrsinoides*, *Acrotriche serrulata* (honey pots), *Hibbertia riparia* (erect guinea-flower), *Lepidosperma filiforme* (common rapiers-sedge). Valley dominated by *L. continentale* ~1 m.
Aireys Inlet	Coastal headland scrub	Coastal cliffs, fragmented vegetation: scrub ~3 m dominated by *Melaleuca lanceolata* (moonah), *Allocasuarina verticillata* (drooping sheoak) *Leptospermum laevigatum* (coast tea-tree); low open woodland, dense understorey *L. continentale*, *L. myrsinoides*, heathland <1.0 m dominated by *Spyridium vexilliferum* (propellor plant), *Leucopogon parviflorus* (coast beard heath), *Lasiopetallum baueri* (slender velvet bush); tussock grassland <0.5 m, *Poa* spp., *Themeda triandra* (kangaroo grass).
Coast dune 36	Heathy woodland, damp heathy mosaic	Dune, interdune scrub. Dense *L. laevigatum*, *Lecopogon parviflorus* ~2 m, understorey *B. marginata*, *P. esculentum*, *Poa* sp., *Leucopogon parviflorus* (coast beard-heath), *Carex apressa* (tall sedge), *Acaena anserinifolia* (bidgee widgee), *Olearia argophylla* (musky daisy bush), *Agrostis billardieri* (coast blown-grass), *Pelargonium australis* (austral storks-bill), *Lepidosperma gladiatum* (coast sword-sedge), *Ficinia nodosa* (knobby club rush).
Coast dune 37	Heathy woodland, damp heathy mosaic	Dune, interdune scrub. Dense *L. laevigatum*, *L. parviflorus* ~2 m, understorey *B. marginata*, *P. esculentum*, *Poa* sp., *L. parviflorus*, *C. apressa*, *A. anserinifolia*, *O. argophylla*, *A billardieri*, *P. australis*, *L.gladiatum*, *F. nodosa*.
Eumeralla	Heathy woodland	Inland south slope. Upper storey *E. obliqua.* Mid–understorey *G. radula*, *L. continentale*, *L. myrsinoides*, *D. glaberrima*, *E. impressa*, *A. serrulata*, *H. riparia*, *Lepidosperma filiforme* (common rapiers-sedge).
Painkalac	Estuarine wetland	Grassland ~1 m of *Poa labillardieri* (common tussock-grass), *L. gladiatum*, *F. nodosa*; herbfields *Phragmites australis* (common reed), *P. esculentum*, *Eleocharis acuta* (common spike-rush), *Juncus kraussii* (sea rush).

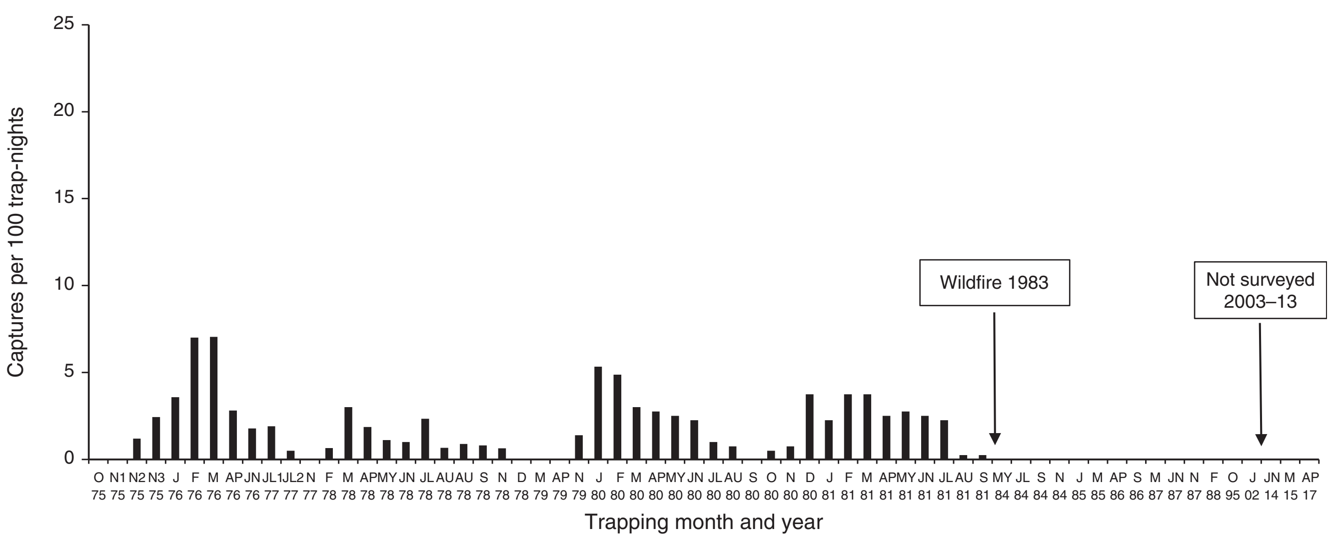

Fig. 2. Trapping success for *A. m. maritimus* in heathy woodland at the Harvey site (1975–2017).

Table 2. Summary of trap success for *A. m. maritimus*, 1975–2007, 2013–17

Status (abundance) change: SD, severe decrease (>80% reduction); D, decrease (30–80% reduction); I, increase. TN, trap-night. References: 1, Wilson *et al.* (1990), Wilson (1983, 1990, 1996); 2, Laidlaw and Wilson (2006), Annett (2008); 3, Gibson *et al.* (2004), Magnusdottir *et al.* (2008), Sale *et al.* (2008); 4, Reichl (1997), Hanley (1999); Wilson *et al.* (2001); 5, Wilson (1990), Aberton (1996), Cullen (2002), Hicks (2004); 6, Homan (2017); 7, Reichl (1997), Hanley (1999)

Site	1975–2007		2013–16		Status (abundance) change (%)	References
	Year last recorded	Maximum trap success per 100 TN	Year last recorded (individuals)	Maximum trap success per 100 TN		
Harvey	1981	7.0	0	0	100 SD	1, 5
Bald Hills B	2005	7.67	2013 (2)	1.33	82 SD	2, 6
Bald Hills C	2005	3.89	2015 (1)	0.33	91 SD	2, 6
Urquhart	2007	20.67	0	0	100 SD	3, 4, 7
Coast dune 36	2001	3.33	2015 (2)	2.22	33 D	4, 5, 7
Coast dune 37	2001	4.44	0	0	100 SD	4, 5, 7
Hutt	1997	1.11	2015 (1[A])	na	99 SD	1, 5
Coast dune 35	1983	2.22	0	0	100 SD	4, 5, 7
Coast dune 39	2001	3.33	0	0	100 SD	4, 5, 7
Eumeralla	2007	2.78	2015 (1)	1.24	45 D	6
Salt Creek 122	2001	2.22	0	0	100 SD	1, 5
Painkalac	2001	13.33	0	0	100 SD	4, 5, 7
Batson 132	1994	4.0[B]	0	0	100 SD	1, 5
Pipeline Pnov 9	1986–88[C]	0.0	2015 (1)	1.11	100 I	1, 5
Aireys Inlet LHW	2002	16.7	0	0	100 SD	4, 5
Aireys Inlet LHE	2002	20.0	0	0	100 SD	4, 5
Aireys Inlet Gully	2003	10.0	0	0	100 SD	4, 5
Aireys Inlet HBB	1999	16.7	0	0	100 SD	4, 5
Edwards Creek	2004	5.00	0[A]	0	100 SD	3
Distillery Creek	2002	5.3	0	0	100 SD	3

[A]Camera records.
[B]Reintroduced animals.
[C]Not recorded.

obtained from the Department of Environment, Land, Water and Planning, Victoria.

Long-term rainfall data were obtained from stations in the eastern Otways at Anglesea and Aireys Inlet (Bureau of Meteorology 2017). Data were used to examine trends in annual and seasonal patterns.

Assessments of site-specific threats

Assessments of threats at *A. m. maritimus* sites were compared for the historical data and recent surveys. An estimation of extent of fragmentation of sites was based on the habitat patch size delineated by roads or tracks. The presence of weeds, of the plant pathogen *P. cinnamomi* and of introduced species (rabbits, dogs, deer) and people were based on site observations, while the presence of predators (fox, cat) was based on camera results and site observations. The degree of the threats were categorised as none, low, medium or high relative to the available data.

Results

Capture summary for A. m. maritimus (2013–17)

Of the 42 sites surveyed since 2013, at 19 of which *A. m. maritimus* was previously recorded, only six sites had a positive record for this species (Table 2). Those sites with a positive trap record had extremely low trapping success, ranging from 0.33 to 2.22 per 100 trap-nights, and exhibited severe declines (>80% reduction) compared with previous years (1975–2007) (Table 2). Seven individual *A. m. maritimus* were caught during the survey trapping; three were captured in 2013–14, four in 2015 and none in 2016 or 2017. Five individuals were captured at targeted sites where *A. m. maritimus* had been recorded previously, two at Bald Hills on Grid B (2013), one on Grid C (2015), and two at coastal dunes Site 36 (2015). No individuals were recorded at six sites that previously had long-term populations, some with very high abundance of *A. m. maritimus*. One individual was recorded at a site where the species had not been recorded previously, Pipeline (2015). Two camera records were at a coastal site (Hutt Gully) where the species had been recorded previously.

Long-term population changes of A. m. maritimus at repeat trapping survey sites

A. m. maritimus was first recorded in heathy woodland at the Harvey site in 1975 and was consistently present for seven years (Fig. 2). Adult males were never trapped after early August, following male 'die off' and total abundance was greatest in January–July in most years. The abundance of the species ranged from 0 to 7.04 captures per 100 trap-nights and density from 1 to 19 animals ha^{-1}. *A. m. maritimus* was not recorded in surveys conducted following the 1983 'Ash Wednesday' fire up to 2002, nor in the recent surveys (2014–17) (Fig. 2, Table 2).

At the Batson site *A. m. maritimus* was first recorded in heathy woodland in 1981, but was not recorded in surveys conducted following the 1983 'Ash Wednesday' fire, up to 1991 (Fig. 3). Trial reintroductions of 10 individuals in 1992, 1993 and 1994 resulted in establishment of the species for three months

(Aberton 1996). Only one individual was captured subsequently, in 1998. The species was absent from the site between 1999 and 2016 (Fig. 3, Table 2).

A. m. maritimus was first recorded at Bald Hills in 2001 (Grid B), and May 2002 (Grid C), 12–13 years after monitoring had begun at these sites (Fig. 4*a*, *b*). The abundance of the species was higher on Grid B (0–7.04 captures per 100 trap-nights) than on Grid C (0–3.89 captures per 100 trap-nights). The density of the population was higher on Grid B (0–8 ha^{-1}) than on Grid C (0–3 ha^{-1}). *A. m. maritimus* was present at maximum abundance from February to August 2002 on Grid B and at lower abundance on both grids from 2003 to 2005. Only female *A. m. maritimus* were trapped on both grids after August, following male 'die off'. The species was captured in very low numbers in recent surveys; two individuals on Grid B in 2013 and one on Grid C in 2015 but none were captured in 2016 or 2017 (Fig. 4*a*, *b*). The populations were present for at least 13 years and exhibited a severe decline (Fig. 4*a*, *b*; Table 2).

A population of *A. m. maritimus* was initially identified at the Urquhart site in 1998 and studies recorded the species consistently from 2001 until 2007 (Fig. 5). Abundance was greatest in January–July in most years, and adult males were never trapped after the middle of July. The species was present at maximum abundance in January 1998 and from January to June 2002. Density estimates varied from a minimum of 1.05 individuals ha^{-1} in November 2004 to a maximum of 27.9 individuals ha^{-1} in January 2002. In 2007 the species was present at lower abundance and no captures were recorded recently (2013–15). The timing of disappearance of the species is unknown due to the gap in surveys between 2007 and 2013. The population was present for at least nine years and exhibited a severe decline (Fig. 5; Table 2).

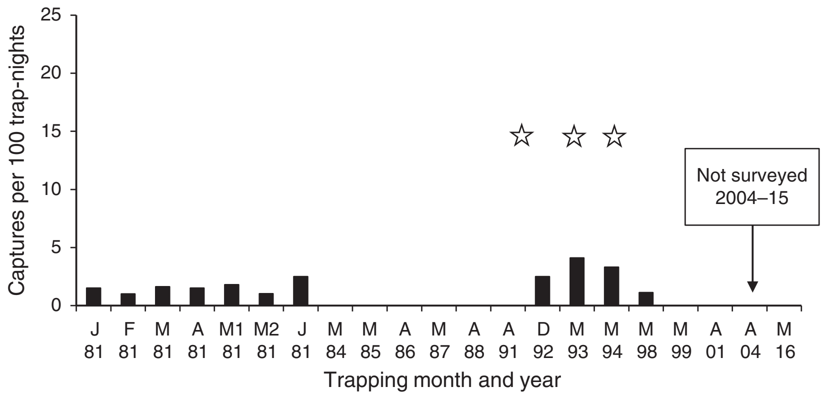

Fig. 3. Trapping success for *A. m. maritimus* in heathy woodland at the Batson site (1981–2016). ☆, reintroductions.

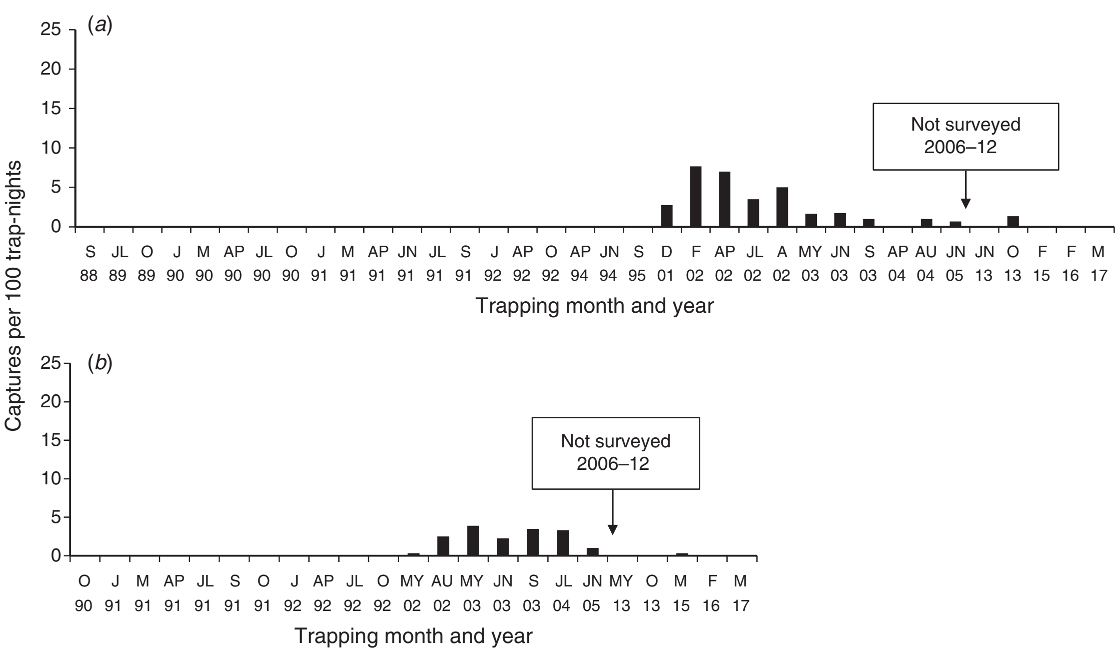

Fig. 4. Trapping success for *A. m. maritimus* in heathy woodland at Bald Hills (1988–2017): (*a*) Grid B, (*b*) Grid C.

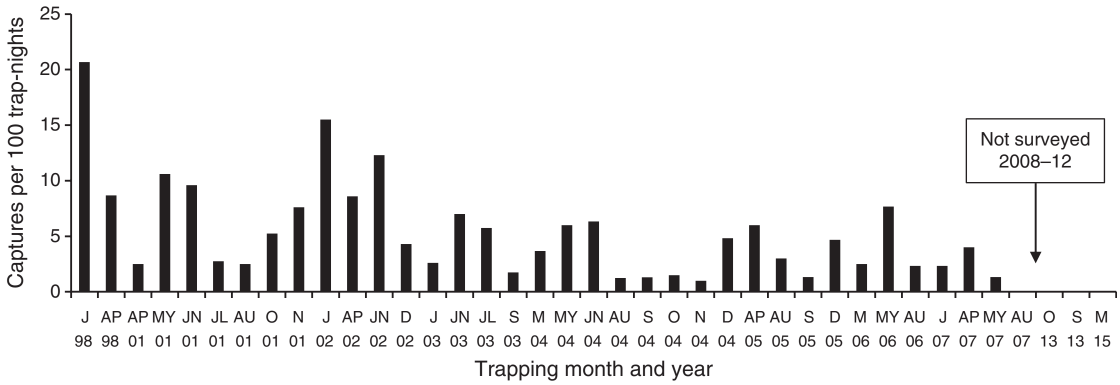

Fig. 5. Trapping success for *A. m. maritimus* in sandy heathland at the Urquhart site (1998–2015).

Table 3. Summary of site-specific threats

Degree of threat: –, none; *, low; **, medium; ***, high. Introduced species: r, rabbit; f, fox; c, cat; d, dog; dr, deer; un, unknown. References: 1, Wilson *et al*. (1990), Wilson (1990, 1996); 2, Laidlaw and Wilson (2006), Annett (2008); 3, Magnusdottir *et al*. (2008), Sale *et al*. (2008); 4, Reichl (1997), Hanley (1999), Wilson *et al*. (2001); 5, Wilson (1990), Aberton (1996), Cullen (2002), Hicks (2004); 6, Homan (2017); 7, Reichl (1997), Hanley (1999)

Site	Wildfire	FRB	Weeds	*Phytophthora*	Introduced species	Fragmentation	Human recreation (dog walking, bike riding) and off-leash dogs	References
Harvey								
1975–2002	**	–	–	*	–	*	–	1, 5
2013–17	–	**	**	**	f**	*	d**	
Batson								
1975–2002	**	–	–	–	–	*	–	1, 5
2013–17	–	–	–	–	–	*	–	
Bald Hills B								
1975–2002	**	*	–	**	f*	*	–	2, 6
2013–17	–	–	–	**	f*	*	–	
Bald Hills C								
1975–2002	**	*	–	**	f*	*	–	2, 6
2013–17	–	–	–	**	f*	*	–	
Urquhart								
1975–2002	**	–	–	–	f **, r*	–	–	3, 4, 7
2013–17	–	–	–	–	f **, r***	–	*	
Aireys Inlet								
1975–2002	**	–	*	–	f *, r*	**	**	4, 5
2013–17	–	–	*	–	f *, r***	***	***	
Coast dunes 36, 37								
1975–2002	**	–	*	–	f *, r*	***	*	4, 5, 7
2013–17	–	–	***	–	f *, c**, r**	***	***	
Hutt								
1975–2002	**	–	*	–	f*, r*	***	–	1, 5
2013–17	–	–	***	–	f*, c*, r*	***	–	
Painkalac								
1975–2002	**	–	*	–	f*, c*	*	**	4, 5, 7
2013–17	–	–	**	–	f*, c*	*	**	
Pipeline								
1975–2002	**	–	–	–	–	–	–	1, 5
2013–17	–	–	–	–	–	–	–	
Eumeralla								
1975–2002	**	–	–	–	un	–	*	6
2013–17	–	–	–	***	f*, dr*	**	***	

At the Eumeralla site a low population of *A. m. maritimus* was recorded between 2004 and 2014. The population exhibited a severe decline with no captures recorded in 2016 (Table 3, Fig. 6).

The earliest record of *A. m. maritimus* in coastal headland scrub at Aireys Inlet was in 1997, where it was present at cliff top sites until 2003 (Fig. 7). The species was present at maximum abundance between 1997 and 1998 but then declined and was not recorded recently (2014–17) (Fig. 7). The population was present for at least six years; however, the timing of disappearance of the species is unknown due to the gap in surveys between 2003 and 2014.

A. m. maritimus was recorded intermittently in coastal sand dune scrub between 1984 and 2001, but was not recorded in 2004. The species was trapped in 2015 but not in 2017 (Fig. 8*a*, *b*; Table 2).

Capture rates of the species in estuarine wetland (Painkalac Creek) were high compared with other the sites (Fig. 9) but the species declined to zero (1999–2001), and it was not recorded in 2015 or 2017.

Threatening processes

Fire history and rainfall

All long-term sites were burnt in the 1983 'Ash Wednesday' wildfire (Table 3). The Harvey site was burnt twice subsequently by fuel reduction burning, and Bald Hills Grids B and C once by fuel reduction burning (Table 3).

The 1983 'Ash Wednesday' wildfire burnt some 40 000 ha regionally and left only small patches of unburnt vegetation in the study area (Wilson *et al*. 1990). Apart from this fire there were no wildfires and little management burning between 1970 and 2005 in the eastern Otways (Department of Environment and Primary Industries 2014). The total area of heathlands burnt by management between 1987 and 2008 (21 years) was 2266 ha (mean = 107.9 ha per annum), and between 2009 and 2012 (3 years) 3095 ha (mean = 1031.67 ha per annum), an annual increase of 900%.

The long-term average rainfall for the eastern Otways was 664 mm (Fig. 10). Record levels of annual deficit rainfall (below average in 11 of 17 years) occurred during the 'millennium drought' (1996–2010) (Fig. 10) (CSIRO and Australian Bureau of Meteorology 2012). Accumulative declines were observed between 1994–99 and 2002–06 during this drought period, with the lowest annual rainfall of 445.8 mm in 2006. High rainfall (900 mm) occurred in 2001, 236 mm above the long-term average. Between 2010 and 2013 rainfall was long-term average; however, rainfall in 2014 (498 mm) and 2015 (448 mm) was exceptionally low, while in 2016 it was above average (Fig. 10).

Site assessments

Prior to 2007 the main processes identified as threatening populations at inland sites were inappropriate fire regimes, *P. cinnamomi* infestation and foxes, while at coastal sites they were habitat fragmentation, foxes and rabbits (Table 3). The processes identified in the recent surveys (2013–17) were similar, with the addition of the presence of dogs, cats, human activity and littering at coastal sites (Table 3). The variability in camera trap deployment may have impacted detection rates of cats and foxes.

Discussion

Current distribution and status of A. m. maritimus in the eastern Otways

The results of our recent surveys (2013–17) that revealed extremely low occurrence and captures of *A. m. maritimus* were entirely unexpected. Only seven individuals were trapped across

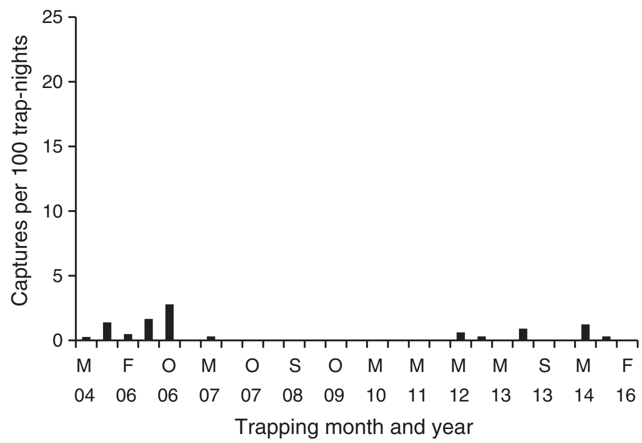

Fig. 6. Trapping success for *A. m. maritimus* in heathy woodland at the Eumeralla site (2004–15).

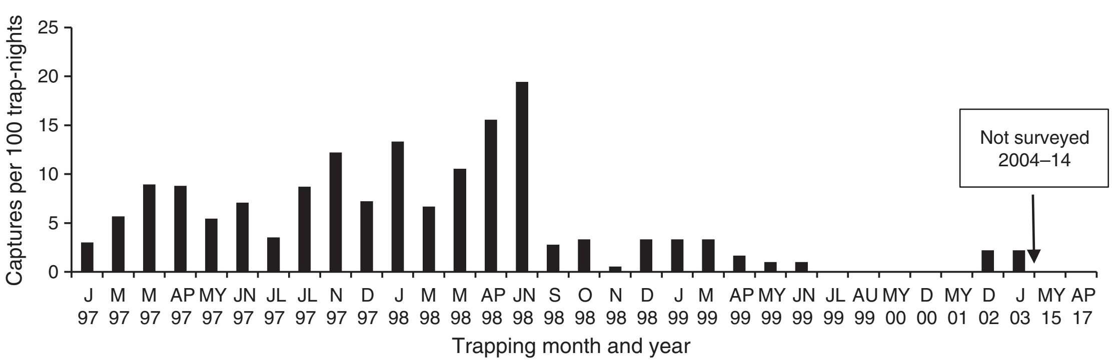

Fig. 7. Trapping success for *A. m. maritimus* in coastal headland scrub at the Aireys Inlet site (1997–2017).

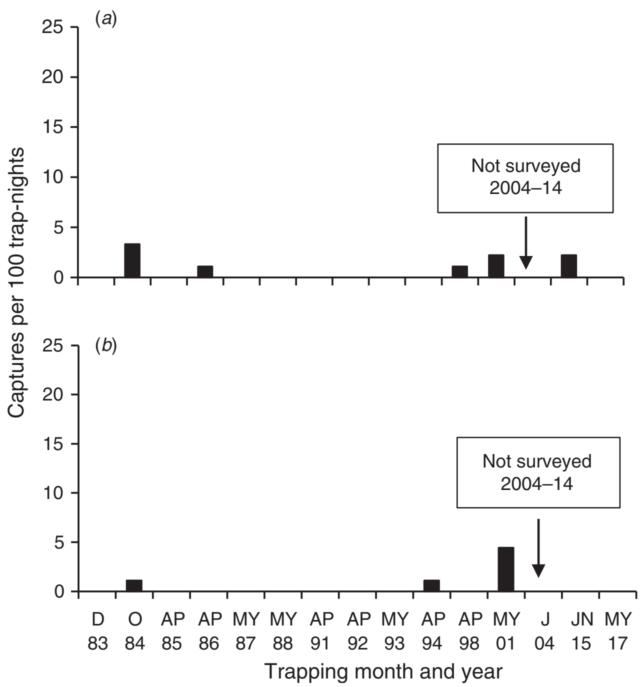

Fig. 8. Trapping success of *A. m. maritimus* in coastal sand dune scrub at (*a*) Site 36 and (*b*) Site 37 (1983–2017).

just five sites. The species was recorded by camera at a sixth site; however, the variability in camera trap deployment may have impacted detection rates of the species. Further, the surveys failed to trap the species at 17 sites where it had been recorded previously, even at sites that once supported high-density or long-term populations. In contrast, *A. m. maritimus* was recorded at 25 sites between 1969 and 2000 (Wilson *et al*. 2001), and at 19 sites in 2001−02 (Gibson *et al*. 2004).

Two of the sites where the species was captured were located in inland heathy woodland communities and two were located coastally. The results are significant because they confirm that the species was surviving until 2013–15 at both inland and coastal sites, albeit at very low abundance. The continued presence (1983–2015) of *A. m. maritimus* in coastal dunes provides evidence that this community may afford significant refuge for the species. The trapping of one individual at an inland site where the species had not been recorded previously may also indicate an important refuge. Although *A. m. maritimus* may still be extant in the eastern Otways, the species has declined significantly in site occupancy and abundance, and may now be represented by only small populations in refugia scattered across the landscape.

While surveys have been completed at 42 sites, assessments are required to evaluate other areas of suitable habitat. A habitat suitability model (Gibson *et al*. 2004) identified the proportion of high-quality suitable habitat (2.8%) in the eastern Otways, representing ~2000 ha. The model provides information to

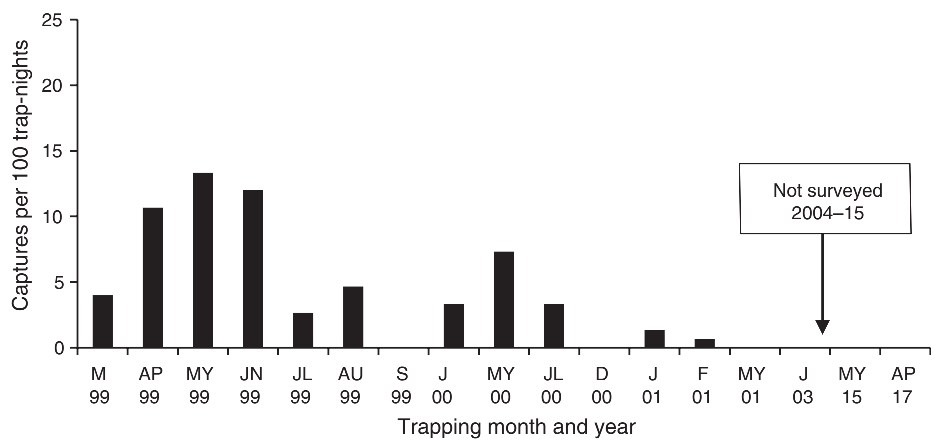

Fig. 9. Trapping success for *A. m. maritimus* in estuarine wetland at Painkalac Creek (1999–2017).

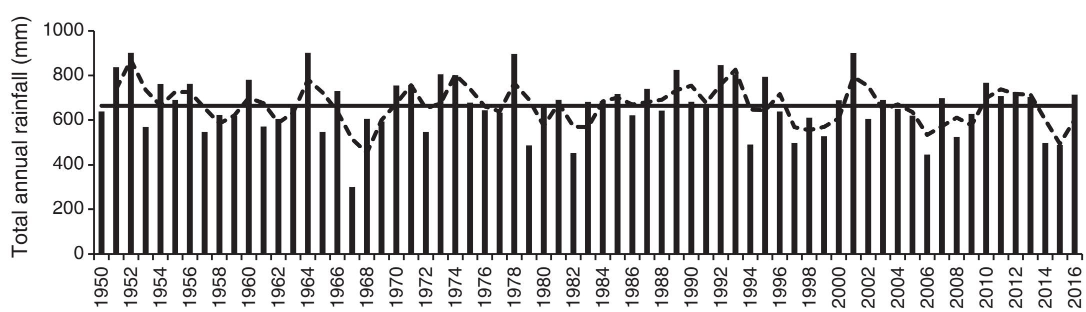

Fig. 10. Total annual rainfall (1950–2016) with moving average (dashed line) and long-term average (solid line).

undertake further surveys to identify any other surviving populations.

Long-term population changes at repeat trapping survey sites

The 10 sites where repeat-measures population data for *A. m. maritimus* is available were established for specific studies into the ecology, life-history and reproduction of the species, not for long-term monitoring purposes. However, they have provided valuable information on the species' population dynamics over time, in different vegetation communities (heathy woodland and sand heathland, coastal headland scrub, coastal sand dune scrub, estuarine wetland), and across the landscape. There was evidence of the persistence of populations of *A. m. maritimus* in heathy woodland and sand heathland over periods of 7–13 years, in coastal headland scrub for 6 years and coastal sand dune scrub for 30 years. The recent trajectories at all sites are similar, providing evidence that there has been a regional decline.

Studies at the Harvey site recorded important information on reproduction and the population dynamics of *A. m. maritimus* over a period of seven years until the 1983 'Ash Wednesday' wildfire (Wilson and Bourne 1984; Wilson 1986; Wilson *et al.* 1986). The species was extirpated by the fire and failed to recolonise between 1984 and 2002, although all other small mammal species did recolonise the site (Wilson *et al*. 1990). The species was also not captured at the site in our recent surveys. Although no trapping was conducted for 10 years (2003–13), fuel reduction burns undertaken at the site in 2003 and 2011 would likely have resulted in extirpation of any individuals if they had recolonised in the interim.

The dynamics of the populations at Urquhart and Bald Hills sites provided important insights into the impacts of rainfall. At the Urquart site peak abundance was recorded in January 2002 following high annual rainfall in 2001 (901 mm), the highest recorded for two decades. The maximum abundance was related to the high survival rates of breeding females (90%) and juveniles following the heavy rainfall, likely due to increased availability of food (Magnusdottir *et al*. 2008). A year later, however, the population had declined to 10% of the peak and precipitation was within the normal range (605–690 mm) in the following three years. The number of young recruited was significantly lower during the spring drought of 2006, in contrast to the wet spring of 2001 (Sale *et al*. 2008). The timing of disappearance of the species from this site is unknown due to the gap in surveys between 2007 and 2013 and no captures were recorded recently (2013–17). *A. m. maritimus* was first recorded at Bald Hills in 2001 but it had been absent for the previous 12 years. The highest captures recorded in late 2001 to 2002 were correlated with high rainfall in the previous 12–21 months and the species was present at lower abundance on both grids from 2003 to 2005 (Annett 2008). The species was captured in low numbers at Bald Hills sites in the recent surveys (2013–15) but none were captured in 2016–17.

Small numbers of *A. m. maritimus* were captured in coastal headland scrub at Airey's Inlet between 1993 and 1996 (Aberton 1996) and the species was consistently present until 2003. High numbers of transient animals at the site were related to the fragmented and narrow nature of the habitat (Wilson *et al*. 2001). The species was not recorded recently (2014–17); however, the timing of disappearance is unknown due to the gap in surveys between 2003 and 2014.

A. m. maritimus was recorded intermittently in coastal dunes between 1984 and 2001. It was trapped in 2015, and two animals were also recorded on camera at the nearby Hutt Gully. The gully is ~100 m from the coastal dunes and the two are connected by constructed drainage that flows under a major road, the Great Ocean Road. The capture of *A. m. maritimus* at these sites is important because it revealed that the species was surviving in these narrow dune systems until recently and may be capable of utilising the under-road drainage to move under a major barrier that fragments its habitat.

The species was first recorded in fringing vegetation of estuarine wetland at Painkalac Creek in 1999, even though survey trapping had been undertaken before this (1990–97) (Aberton 1996; Reichl 1997). The population increased in May 2000 before decreasing to zero in December 2000. No *A. m. maritimus* were present thereafter (2001–03) and the species was not recorded in 2015 or 2017. Foxes and cats have been recorded in these sites and pose a major threat.

The densities of *A. m. maritimus* populations in the eastern Otways varied considerably: from 27.9 individuals ha^{-1} (Urquhart) to 1.1 individuals ha^{-1} at the Aireys Inlet site (Wilson *et al*. 2001; Annett 2008; Magnusdottir *et al*. 2008; Sale *et al*. 2008). While the maximum density at the Urquhart site was related to high rainfall, the lowest density at Aireys Inlet is a reflection of the narrow and fragmented habitat (Wilson *et al*. 2001; Annett 2008; Magnusdottir *et al*. 2008; Sale *et al*. 2008).

Declines and refuges

The recording of *A. m. maritimus* in the coastal dunes (2015) is significant because it provides evidence of survival of the species in the dune system over a 30-year period, albeit in low numbers. Our recent surveys also found these systems to be species rich ($n = 5$), with high mammal capture rates (20–47%) particularly for the rodents *Rattus fuscipes* and *R. lutreolus* (Wilson and Garkaklis 2016). Historical data (1983–2004) show that the dunes have consistently supported between five and seven native species, and high-density populations (Wilson *et al*. 1990; Aberton 1996; Wilson and Garkaklis 2016).

Survival of *A. m. maritimus* and high-density populations of other small mammals in the dunes may be due to mammals benefiting indirectly from marine inputs such as seabird guano and other allochthonous marine nutrient inputs at these sites (Bancroft *et al*. 2005). Increased nutrient inputs have been linked to high food availability (invertebrates), dense vegetation cover, higher abundance of wildlife and increased reproductive success in island and coastal sites (Wolfe *et al*. 2004; Bancroft *et al*. 2005; Sale *et al*. 2008; Sale and Arnould 2012). Very-high-density populations of *A. m. maritimus* that have been recorded on island habitats have also been shown to be supported in this way (Sale *et al*. 2008; Sale and Arnould 2012). We have observed seabirds inhabiting these dune sites, together with seabird and seal carcasses.

In the recent surveys *A. m. maritimus* was also recorded at an inland site. The site had been trapped annually since 1984 and the open heathy understorey that developed there after the 1983

'Ash Wednesday' wildfire has been replaced, over a thirty-year period, with dense vegetation cover dominated by *Leptospermum continentale* and *Gahnia radula* at <1 m and *Hakea decurrens* up to 2 m in height. In the absence of fire the changes have resulted in structurally complex habitat that is optimal for *A. m. maritimus* (Moro 1991; Aberton 1996; Wilson *et al*. 2001; Gibson *et al*. 2004). These changes provide evidence of substantial long-term structural transformations that can occur in habitats in the eastern Otways in the absence of fire. There is a need to identify the current location and extent of both the dune and inland optimal habitat across the current landscape, as they represent potential refuges for the species.

Impacts of threats affecting decline of A. m. maritimus in the eastern Otways

While several factors contributing to the decline of *A. m. maritimus* have been identified the relative impacts and their interactions in the eastern Otways are complex, and not well understood (Wilson *et al.* 1990, 2001; Laidlaw and Wilson 2006; Magnusdottir *et al*. 2008; Sale *et al*. 2008; Wilson and Garkaklis 2016). Habitat clearing originally for farming and subsequently from the 1960s for coal mining (~600 ha), have likely contributed to fragmentation of *A. m. maritimus* habitat. The current degree of fragmentation at *A. m. maritimus* sites varies, with coastal sites being highly fragmented compared with inland sites (Table 3). This study found that populations at Aireys Inlet that were previously identified to be at high risk from fragmentation impacts (Wilson *et al*. 2001) are now no longer extant. Fragmentation is of particular importance in this species as females are philopatric and males dispersive (Magnusdottir *et al*. 2008). High fragmentation levels may thus restrict male dispersal between the home ranges of females, affecting mating success and increasing exposure to predation when dispersing through unsuitable habitat. High levels of transient animals have been recorded for several populations (Aireys Inlet, Urquhart), compared with other non-fragmented sites (Wilson *et al*. 2001; Magnusdottir *et al*. 2008).

A. m. maritimus is a late-successional species (Wilson *et al.* 2001; Gibson *et al*. 2004) and the impact of too frequent burning is recognised as a significant threat (Woinarski *et al*. 2014; Threatened Species Scientific Committee 2016). Eight of our *A. m. maritimus* long-term study sites were last burnt in the 1983 wildfire and support late-successional complex habitat (Wilson *et al*. 2001; Gibson *et al*. 2004). Although the locations where the species was recorded may not have been burnt for decades the surrounding area may have been burnt multiple times and thus be unsuitable for *A. m. maritimus* to occupy or traverse. This can result in fragmentation of suitable habitat and isolation of the species in small areas of optimal aged vegetation. The increase of 900% in annual mean burning of the heathlands in 2009–12 is of great concern for this species and has implications for the availability of optimal late-successional habitat (Wilson 1991; Wilson *et al*. 2001).

While low rainfall was proposed as a likely factor contributing to declines in *A. m. maritimus* populations in the eastern Otways (Wilson *et al*. 2001) there is now strong evidence from several sites (e.g. Urquharts, Bald Hills) that rainfall does have major impacts on population dynamics (Annett 2008; Magnusdottir *et al*. 2008; Sale *et al*. 2008). Maximum population densities occurred following the highest total annual rainfall (901 mm) recorded for two decades, and density declines were measured during periods of below-average rainfall and drought (2001–07). The impact of rainfall on the species is considered to result from bottom-up increases or declines in productivity of vegetation and associated dietary resources, particularly moth larvae and beetles (Sale *et al*. 2008; Sale and Arnould 2012). The increase in survival of females into the breeding season, of juveniles after weaning, and of overall body weight following peak annual rainfall has been related to increased productivity while lower rainfall and drought result in decreased productivity with consequential population declines.

Declining rainfall during the 'millennium drought' (1996–2010), and exceptionally low rainfall in 2014 (498 mm) and 2015 (448) is likely to have impacted *A. m. maritimus* across the landscape. The high rainfall in July 2016 followed by high spring rainfall was expected to result in increased survival and recruitment. There was no increase in locations or trap success of *A. m. maritimus* in 2017, providing no evidence of such responses. This indicates that recovery may require longer periods of increased rainfall.

Several threats are likely to have complex synergistic interactions on the population dynamics of *A. m. maritimus*. For example, wildfire or fuel reduction management burns during drought conditions increase the risk of extirpation of small fragmented populations and decrease recovery opportunities. The recent evidence that the impacts of introduced predators on mammals are more severe in burnt areas (McGregor *et al.* 2014, 2015; Leahy *et al*. 2015; Hradsky *et al*. 2017) indicates that the impact of predators will be exacerbated under increased burning regimes. There is a need to understand further how these factors interact in order to develop recovery and management plans under predicted climate change.

Management implications for A. m. maritimus in a drying climate

Climate change involving lower rainfall and extended droughts poses significant threats to this vulnerable species. The projected rainfall decrease over the area under high carbon emission scenarios is 25–45% by 2090 (Hope *et al*. 2015). While it is not possible to avoid predicted rainfall declines there is an opportunity to mitigate current threats and increase resilience of the species with appropriate management such as implementation of suitable fire regimes, establishment and protection of refuges and ecological linkages and control of predators.

An increase in fuel reduction burning has been implemented across the region since 2009 (Parliament of Victoria 2010; Department of Environment and Primary Industries 2014). In 2013, 42% of the Anglesea heathland was below its 'minimum tolerable fire interval', and 37% was at an early growth stage (Department of Environment and Primary Industries 2014). This has significant implications for the availability of optimal habitat for *A. m. maritimus*, which is dependent on structurally complex, late-successional habitat (Wilson *et al*. 1986; Moro 1991; Wilson 1996; Gibson *et al*. 2004). Implementation of burning regimes to ensure access to appropriate age classes need to be given high priority. Detailed spatial fire data are required to inform optimum burning regimes and to maintain suitable successional aged habitat across the landscape for survival after fire, dispersal and recolonisation (Bradstock *et al*. 2005; Clarke 2008; MacHunter *et al*. 2009; Driscoll *et al*. 2010).

While there is no direct evidence of impacts on *A. m. maritimus*, there is evidence of fox and cat predation on mammals in the eastern Otways and populations of the species located in isolated and fragmented habitat are likely to be very susceptible to predation (Hutchings 1996, 2003; Wilson and Wolrige 2000; Wilson *et al*. 2001; Hradsky *et al*. 2017). There is no targeted cat control in the region. While fox baiting in the region occurs four times a year (Parks Victoria 2015), the effectiveness of the current baiting program for *A. m. maritimus* needs reviewing. In addition, integrated management of predators and fire is strongly recommended (Doherty *et al*. 2015).

Providing key refuge sites can buffer a species from several threats. It is crucial that conservation efforts focus on refuge areas for *A. m. maritimus*. It is recommended that refuge areas such as coastal dunes be identified across the landscape, utilising current or improved habitat models and remote-sensing techniques (Ferrier 2012), and that intensive management (protection from inappropriate fire, control of rabbits, introduced predators, dogs) be implemented as a priority.

The long-term mammal research programs in this study were not established as monitoring programs, so were without defined population condition break-points that would prompt management responses (Burgman *et al*. 2012). There is a need to implement consistent monitoring of *A. m. maritimus* that provides effective measures of declines, for management. Reporting on the effectiveness of fire management on public land to provide evidence that burning is delivering optimal habitat for this species is also required (Department of Environment, Land, Water and Planning 2015).

Overall status of A. m. maritimus – effective management and monitoring

A. m. maritimus was considered to be at Lower Risk (near threatened) (Menkhorst 1995). Although the species has recently been listed as Vulnerable (*EPBCA 1999*) (Threatened Species Scientific Committee 2016) this study has provided strong evidence of continued decline of populations, extent of occurrence and area of occupancy in the eastern Otways. There is also evidence that the subspecies' distribution declined significantly throughout Victoria between 1990 and 2014 (Threatened Species Scientific Committee 2016). High-priority conservation actions that we have recommended for *A. m. maritimus* need to be applied throughout its range, together with an integrated monitoring program across its range linked to an assessment of management effectiveness.

Conflicts of interest

The authors declare no conflicts of interest.

Acknowledgements

We gratefully acknowledge our colleagues, postgraduates and undergraduates for their contributions to the long-term studies and their continued support, including: Dr J. Aberton, Dr K. Annett, Dr M. Antos (Parks Vic), Dr M. Cullen, E. Danby (Parks Vic), A. Hanley, Dr K. Hicks, Dr M. Ibbett (DELWP), Dr R. Jessop, Dr W. S. Laidlaw, A. Williamson. Many thanks to volunteers (A. Garkaklis, J. Campbell, D. Hopper, J. Hopper, J. Naylor-Pratt) for their assistance in the field. Financial support from the Australian Research Council, Alcoa Australia, Deakin University, Deakin Foundation, Holsworth Wildlife Research Endowment, M. A. Ingram Trust, and Parks Victoria is gratefully acknowledged. All work between 2013 and 2017 was conducted under the Victorian Wildlife Act and National Parks Act, and scientific permits approved by the Department of Environment, Land, Water and Planning and Parks Victoria, under approval of the Deakin University Animal Ethics Committee (B05-2013, B01-2016).

References

Aberton, J. G. (1996). Succession of small mammal communities after fire and reintroduction of the swamp antechinus, *Antechinus minimus*. Ph.D. Thesis, Deakin University, Geelong.

Allison, L. M., Gibson, L. A., and Aberton, J. G. (2006). Dietary strategy of the swamp antechinus (*Antechinus minimus maritimus*) (Marsupialia: Dasyuridae) in coastal and inland heathland habitats. *Wildlife Research* **33**, 67–76. doi:10.1071/WR05038

Annett, K. (2008). Impacts of the plant pathogen *P. cinnamomi* (cinnamon fungus) on fauna and ecosystem functions. Ph.D. Thesis, Deakin University, Geelong.

Bachmann, M., and van Weenen, J. (2001). Distribution and status of the swamp antechinus (*Antechinus minimus maritimus*) (Marsupialia: Dasyuridae) in South Australia. The Nature Conservation Society of South Australia Inc & Department for Environment and Heritage, Adelaide.

Bancroft, W. J., Roberts, J. D., and Garkaklis, M. J. (2005). Burrowing seabirds drive decreased diversity and structural complexity, and increased productivity in insular-vegetation communities. *Australian Journal of Botany* **53**, 231–241. doi:10.1071/BT04079

Bilney, R. J., Cooke, R., and White, J. G. (2010). Underestimated and severe: small mammal decline from the forests of south-eastern Australia since European settlement, as revealed by a top-order predator. *Biological Conservation* **143**, 52–59. doi:10.1016/j.biocon.2009.09.002

Bradstock, R. A., Bedward, M., Gill, A. M., and Cohn, J. S. (2005). Which mosaic? A landscape ecological approach for evaluating interactions between fire regimes, habitat and animals. *Wildlife Research* **32**, 409–423. doi:10.1071/WR02114

Burbidge, A. A., and McKenzie, N. L. (1989). Patterns in the modern decline of Western Australia's vertebrate fauna: causes and conservation implications. *Biological Conservation* **50**, 143–198. doi:10.1016/0006-3207(89)90009-8

Burbidge, A. A., McKenzie, N. L., Brennan, K. E. C., Woinarski, J. C. Z., Dickman, C. R., Baynes, A., Gordon, G., Menkhorst, P. W., and Robinson, A. C. (2008). Conservation status and biogeography of Australia's terrestrial mammals. *Australian Journal of Zoology* **56**, 411–422. doi:10.1071/ZO08027

Bureau of Meteorology (2017). Climate data on-line. Commonwealth of Australia. Available at: www.bom.gov.au/climate

Burgman, M., Lowell, K., Woodgate, P., Jones, S., Richards, G., and Addison, P. (2012). An endpoint hierarchy and process control charts for ecological monitoring. In 'Biodiversity Monitoring in Australia'. (Eds D. Lindenmayer and P. Gibbons.) pp. 71–78. (CSIRO Publishing: Melbourne.)

Claridge, A. W., Paull, D. J., Simon, C., and Barry, S. C. (2010). Detection of medium-sized ground-dwelling mammals using infrared digital cameras: an alternative way forward? *Australian Mammalogy* **32**, 165–171. doi:10.1071/AM09039

Clarke, M. F. (2008). Catering for the needs of fauna in fire management: science or just wishful thinking? *Wildlife Research* **35**, 385–394. doi:10.1071/WR07137

Coates, F., Loyn, R., Ibbett, M., and Brown, G. (2013). Progress report, Otways Hawkeye: biodiversity monitoring for improved fire management. Arthur Rylah Institute for Environmental Research, Unpublished report. Department of Environment and Primary Industries, Melbourne.

CSIRO and Australian Bureau of Meteorology (2012). State of the Climate 2012. Available at: www.climatechangeinaustralia.gov.au

CSIRO and Australian Bureau of Meteorology (2014). State of the Climate 2014. Available at: www.climatechangeinaustralia.gov.au

CSIRO and Australian Bureau of Meteorology (2015). Climate change in Australia. Technical Report. CSIRO and Australian Bureau of Meteorology.

Cullen, M. J. (2002). Long-term temporal changes in small mammal communities in the eastern Otway Ranges, Australia. B.Sc. (Honours) Thesis, Deakin University, Geelong.

De Bondi, N., White, J. G., Stevens, M., and Cooke, R. (2010). A comparison of the effectiveness of camera trapping and live trapping for sampling terrestrial small-mammal communities. *Wildlife Research* **37**, 456–465. doi:10.1071/WR10046

Department of Environment, Land, Water and Planning (2015). Monitoring, evaluation and reporting framework for bushfire management on public land. State of Victoria, Melbourne.

Department of Environment and Primary Industries (2014). Strategic bushfire management plan. Barwon Otway bushfire risk landscape. State of Victoria, Melbourne.

Doherty, T. S., Chris, R., Dickman, C. R., Dale, G., Nimmo, D. G., and Ritchie, E. G. (2015). Multiple threats, or multiplying the threats? Interactions between invasive predators and other ecological disturbances. *Biological Conservation* **190**, 60–68. doi:10.1016/j.biocon.2015.05.013

Driscoll, D. A., Lindenmayer, D. B., Bennett, A. F., Bode, M., Bradstock, R. A., Cary, G. J., Clarke, M. F., Dexter, N., Fensham, R., Friend, G., Gill, M., James, S., Kay, G., Keith, D. A., MacGregor, C., Russell-Smith, J, Salt, D, Watson, J. E. M., Williams, R. J., and York, A (2010). Fire management for biodiversity conservation: key research questions and our capacity to answer them. *Biological Conservation* **143**, 1928–1939. doi:10.1016/j.biocon.2010.05.026

Ferrier, S. (2012). Big-picture assessment of biodiversity change: scaling up monitoring without selling out on scientific rigour. In 'Biodiversity Monitoring in Australia'. (Eds D. Lindenmayer and P. Gibbons.) pp. 63–70. (CSIRO Publishing: Melbourne.)

Gibson, L. A., Wilson, B. A., Cahill, D. M., and Hill, J. (2004). Modelling habitat suitability of the swamp antechinus (*Antechinus minimus maritimus*) in the coastal heathlands of southern Victoria, Australia. *Biological Conservation* **117**, 143–150. doi:10.1016/S0006-3207(03)00288-X

Hanley, A. (1999). A comparison of the population ecology of *Antechinus minimus maritimus* in fragmented and contiguous habitats at Aireys Inlet, Victoria. B.Sc.(Honours) Thesis, Deakin University, Melbourne.

Hicks, K. (2004) Modelling the distribution of small mammals in the Eastern Otway Ranges. B.Sc.(Honours) Thesis, Deakin University, Geelong.

Homan, P. (2017). Surveys of vertebrate fauna of the Eumeralla section of the Great Otway National Park, Victoria 2004–2015. I. Mammals. *Victorian Field Naturalist* **134**, 11–18.

Hope, P., Grose, M. R., Timbal, B., Dowdy, A. J., Bhend, J., Katzfey, J. J., Bedin, T., Wilson, L., and Whetton, P. H. (2015). Seasonal and regional signature of the projected southern Australian rainfall reduction. *Australian Meteorological and Oceanographic Journal* **65**, 54–71. doi:10.22499/2.6501.005

Hradsky, B. A. K., Mildwaters, C., Ritchie, E. G., Christie, F., and Di Stefano, J. (2017). Responses of invasive predators and native prey to a prescribed forest fire. *Journal of Mammalogy* **98**, 835–847. doi:10.1093/jmammal/gyx010

Hutchings, S. D. (1996). The effects of a fox control program on small & medium sized mammals in the Anglesea Heathlands, Victoria. M.Sc. Thesis, Deakin University, Geelong.

Hutchings, S. (2003). Behaviour & ecology of cats in Anglesea heathlands, Victoria. Ph.D. Thesis, Deakin University, Geelong.

Johnson, C. (2006). 'Australia's Mammal Extinctions: A 50,000 Year History.' (Cambridge University Press: Melbourne.)

Laidlaw, W. S., and Wilson, B. A. (2006). Habitat utilisation by small mammals in a coastal heathland exhibiting symptoms of *Phytophthora cinnamomi* infestation. *Wildlife Research* **33**, 639–649. doi:10.1071/WR05080

Land Conservation Council Victoria (1985). Report of the Land Conservation Council for the year ended 30 June 1985: ordered by the Legislative Assembly to be printed. F.D. Atkinson, Government Printer, Melbourne.

Leahy, L., Legge, S. M., Tuft, K., McGregor, H., Barmuta, L., Jones, M. E., and Johnson, C. N. (2015). Amplified predation after fire suppresses rodent populations in Australia's tropical savannas. *Wildlife Research* **42**, 705–716. doi:10.1071/WR15011

Lindenmayer, D. B., Gibbons, P., Bourke, M., Burgman, M., Dickman, C. R., Ferrier, S., Fitzsimons, J., Freudenberger, D., Garnett, S. T., Groves, C., Hobbs, R. J., Kingsford, R. T., Krebs, C., Legge, S., Lowe, A. J., McLean, R., Montambault, J., Possingham, H., Radford, J., Robinson, D., Smallbone, L., Thomas, D., Varcoe, T., Vardon, M., Wardle, G., Woinarski, J., and Zerger, A. (2012). Improving biodiversity monitoring. *Austral Ecology* **37**, 285–294. doi:10.1111/j.1442-9993.2011.02314.x

Lindenmayer, D., Burns, E., Thurgate, N., and Lowe, A. (2014) 'Biodiversity and Environmental Change: Monitoring, Challenges and Direction.' (CSIRO Publishing: Melbourne.)

MacHunter, J., Menkhorst, P., and Loyn, R. (2009). Towards a process for integrating vertebrate fauna into fire management planning. Arthur Rylah Institute for Environmental Research. Victorian Government, Melbourne.

Magnusdottir, R., Wilson, B. A., and Hersteinsson, P. (2008). Dispersal and the influence of rainfall on a population of swamp antechinus (*Antechinus minimus maritimus*). *Wildlife Research* **35**, 446–454. doi:10.1071/WR06156

Maxwell, S., Burbidge, A. A., and Morris, K. (1996). 'Action Plan for Australian Marsupials and Monotremes.' (Wildlife Australia: Canberra.)

McGregor, H. W., Legge, S., Jones, M. E., and Johnson, C. N. (2014). Landscape management of fire and grazing regimes alters the fine-scale habitat utilisation by feral cats. *PLoS One* **9**, e109097. doi:10.1371/journal.pone.0109097

McGregor, H., Legge, S., Jones, M. E., and Johnson, C. N. (2015). Feral cats are better killers in open habitats, revealed by animal-borne video. *PLoS One* **10**, e0133915. doi:10.1371/journal.pone.0133915

McKenzie, N. L., Burbidge, A., Baynes, A., Brereton, R. N., Dickman, C. R., Gordon, G., Gibson, L. A., Menkhorst, A. C., Robinson, A. C., Williams, M. R., and Woinarski, J. C. Z. (2007). Analysis of factors implicated in the recent decline of Australia's mammal fauna. *Journal of Biogeography* **34**, 597–611. doi:10.1111/j.1365-2699.2006.01639.x

McMahon, K., and Brighton, M. (2002). 'Anglesea Heath Management Plan November 2002.' (Parks Victoria: Melbourne.)

Menkhorst, P. (1995). 'Mammals of Victoria. Distribution, Ecology and Conservation.' (Oxford University Press: Melbourne.)

Moro, D. (1991). The distribution of small mammal species in relation to heath vegetation near Cape Otway, Victoria. *Wildlife Research* **18**, 605–618. doi:10.1071/WR9910605

Parks Victoria (2015). Eastern Otways Fox Control Program report (2011–2015). Parks Victoria, Anglesea.

Parks Victoria and Department of Sustainability and Environment (2009). Caring for country: the Otways and you. Great Otway National Park and Otway Forest Park Management Plan. Parks Victoria and DSE, Melbourne.

Parliament of Victoria (2010). 2009 Victorian Bushfires Royal Commission: final report; summary. Parliament of Victoria, Melbourne.

Reichl, T. (1997). The effects of habitat fragmentation on *Antechinus minimus* the swamp antechinus, in the eastern Otway Ranges, Victoria. B.Sc. (Honours) Thesis, Deakin University, Melbourne.

Sale, M. G., and Arnould, J. P. Y. (2012). Inflated population density of island antechinus: a case of allochthonous marine inputs leading to increased food availability? *Australian Journal of Zoology* **60**, 343–351. doi:10.1071/ZO12073

Sale, M. G., Ward, S. J., and Arnould, J. P. Y. (2006). Aspects of the ecology of swamp antechinus (*Antechinus minimus maritimus*) on a Bass Strait island. *Wildlife Research* **33**, 215–221. doi:10.1071/WR05051

Sale, M. G., Wilson, B. A., and Arnould, J. P. Y. (2008). Factors influencing population dynamics in island and mainland populations of swamp antechinus (*Antechinus minimus*; Marsupialia). *Australian Journal of Zoology* **56**, 187–194. doi:10.1071/ZO08076

Threatened Species Scientific Committee (2016). Conservation advice for listing of *Antechinus minimus maritimus* (swamp antechinus). *Environment Protection and Biodiversity Conservation Act 1999*. Commonwealth of Australia, Canberra.

Van Dyck, S., and Strahan, R. (2008). 'The Complete Book of Australian Mammals.' (New Holland Publishers: Brisbane.)

Van Dyck, S., Gynther, I., and Baker, A. (2013). 'Field Companion to The Mammals of Australia.' (New Holland Publishers: Brisbane.)

van Weenen, J., and Menkhorst, P. (2008). *Antechinus minimus*. In 'IUCN Red List of Threatened Species'. Version 2012.1. Available at: www.iucnredlist.org [accessed 5 April 2017].

Varcoe, T. (2012). A park manager's perspective on ecological monitoring. In 'Biodiversity Monitoring in Australia'. (Eds D. Lindenmayer and P. Gibbons.) pp. 101–110. (CSIRO Publishing: Melbourne.)

Wayne, A. F., Wilson, B. A., and Woinarski, J. C. Z. (2017). Falling apart? Insights and lessons from three recent studies documenting rapid and severe decline in terrestrial mammal assemblages of northern, south-eastern and southwestern Australia. *Wildlife Research*, in press. doi:10.1071/WR16178

Wilson, B. A. (1983). Reproduction and aspects of ecology of *Antechinus minimus maritimus*. M.Sc. Thesis, La Trobe University, Melbourne.

Wilson, B. A. (1986). Reproduction in the female dasyurid *Antechinus minimus maritimus* (Marsupialia: Dasyuridae.) *Australian Journal of Zoology* **34**, 189–197. doi:10.1071/ZO9860189

Wilson, B. A. (1990). The effects of vegetation, fire and other disturbance factors on small mammal ecology and conservation. Ph.D. Thesis, Deakin University, Geelong.

Wilson, B. A. (1991). The ecology of *Pseudomys novaehollandiae* (Waterhouse 1843) in the eastern Otway Ranges, Victoria. *Wildlife Research* **18**, 233–247. doi:10.1071/WR9910233

Wilson, B. A. (1996). Fire effects on vertebrate fauna and implications for fuel reduction burning and management. In 'Fire and Biodiversity: the Effects and Effectiveness of Fire Management. Proceedings of the Biodiversity Series Conference, Footscray, Melbourne, 8–9 October 1994'. (Ed. Victorian National Association.) pp. 131–147. (Biodiversity Unit, Department of Environment, Sport and Territory: Canberra.)

Wilson, B. A., and Bachmann, M. R. (2008). *Antechinus minimus* (swamp antechinus). In 'The Complete Book of Australian Mammals'. (Eds S. Van Dyck and R. Strahan.) pp. 93–94. (New Holland Publishers: Brisbane.)

Wilson, B. A., and Bourne, A. R. (1984). Reproduction in the male dasyurid *Antechinus minimus maritimus* (Marsupialia, Dasyuridae.) *Australian Journal of Zoology* **32**, 311–318. doi:10.1071/ZO9840311

Wilson, B. A., and Garkaklis, M. J. (2016). Assessment of the status and distribution of the endangered New Holland mouse and the swamp antechinus in the eastern Otways: data for the Parks Victoria Decision Support System. Unpublished report. Parks Victoria, Melbourne.

Wilson, B. A., and Wolrige, J. (2000). Assessment of the diet of the fox, *Canis vulpes* in habitats of the eastern Otway Ranges, Victoria. *Australian Mammalogy* **22**, 201–211.

Wilson, B. A., Bourne, A. R., and Jessop, R. E. (1986). Ecology of small mammals in coastal heathland at Anglesea, Victoria. *Australian Wildlife Research* **13**, 397–406. doi:10.1071/WR9860397

Wilson, B. A., Robertson, D., Moloney, D. J., Newell, G. R., and Laidlaw, W. S. (1990). Factors affecting small mammal distribution and abundance in the eastern Otway Ranges, Victoria. *Proceedings of the Ecological Society of Australia* **16**, 379–396.

Wilson, B. A., Aberton, J., and Reichl, T. (2001). Effects of fragmented habitat and fire on the distribution and ecology of the swamp antechinus (*Antechinus minimus maritimus*) in the eastern Otways, Victoria. *Wildlife Research* **28**, 527–536. doi:10.1071/WR00016

Wilson, B. A., Dickman, C., and Fletcher, T. (2003). Dasyurid dilemmas: problems and solutions for conserving Australia's small carnivorous marsupials. In 'Predators with Pouches: The Biology of Carnivorous Marsupials'. (Eds M. E. Jones, C. R. Dickman, and M. Archer.) pp. 403–417. (CSIRO Publishing: Melbourne.)

Woinarski, J. C. Z., Burbidge, A. A., and Harrison, P. L. (2014). 'The Action Plan for Australian Mammals 2012.' (CSIRO Publishing: Melbourne.)

Woinarski, J. C. Z., Burbidge, A. A., and Harrison, P. L. (2015). Ongoing unravelling of a continental fauna: decline and extinction of Australian mammals since European settlement. *Proceedings of the National Academy of Sciences of the United States of America* **112**, 4531–4540. doi:10.1073/pnas.1417301112

Wolfe, K. M., Mills, H. R., Garkaklis, M. J., and Bencini, R. (2004). Post-mating survival in a small marsupial is associated with nutrient input from seabirds. *Ecology* **85**, 1740–1746. doi:10.1890/03-3091

Handling Editor: Janine Deakin